ABSTRACT

Title of Dissertation: SECRETIVE MARSHBIRDS OF URBAN WETLANDS IN THE WASHINGTON, DC METROPOLITAN AREA

Patrice Nielson, Doctor of Philosophy 2016

Dissertation directed by: Dr. William Bowerman and
Dr. Andrew Baldwin
Environmental Science and Technology

Secretive marshbirds are in decline across their range and are species of greatest conservation need in state Wildlife Action Plans. However, their secretive nature means there is relatively sparse information available on their ecology. There is demand for this information in the Washington, DC area for updating conservation plans and guiding wetland restoration. Rapid Wetland Assessment Methods are often used to monitor success of restoration but it is unknown how well they indicate marshbird habitat. Using the Standardized North American Marshbird Monitoring Protocol, I surveyed 51 points in 25 marshes in the DC area in 2013 – 2015. I also collected data on marsh area, buffer width, vegetation/water interspersion, vegetation characteristics, flooding, and invertebrates. At each bird survey point I assessed wetland quality using the Floristic Quality Assessment Index (FQAI) and California Rapid Wetland Assessment (CRAM) methods. I used Program Presence to model detection and occupancy probabilities of secretive marshbirds as a function of habitat variables. I found king rails (*Rallus elegans*)

at five survey sites and least bittern (*Ixobrychus exilis*) at thirteen survey sites. Secretive marshbirds were using both restored and natural marshes, marshes with and without invasive plant species, and marshes with a variety of dominant vegetation species. King rail occupancy was positively correlated with plant diversity and invertebrate abundance and weakly negatively correlated with persistent vegetation. Least bittern occupancy was strongly negatively correlated woody vegetation and invertebrate abundance and weakly positively correlated with persistent vegetation. Species-specific models provided a better fit for the data than generic marshbird models. A comparison model based on important habitat variables in other regions was a very poor fit for the data in all sets of models tested. FQAI was a better indicator of secretive marshbird presence than CRAM, but neither method had very good predictive ability or goodness of fit. These results underscore the importance of having species- and region-specific models for effective conservation. Based on these findings, decreasing woody vegetation and managing for a variety of co-dominant species to avoid monocultures would improve habitat for marshbirds. Rapid Assessment Method scores should be interpreted with caution when applied to marshbird habitat conservation.

PRESENCE AND HABITAT REQUIREMENTS OF SECRETIVE MARSHBIRDS IN THE WASHINGTON, DC METROPOLITAN AREA

by

Patrice Nielson

Dissertation submitted to the Faculty of the Graduate School of the
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Advisory Committee:
Professor Andrew Baldwin, Co-Chair
Professor William Bowerman, Co-Chair
Professor Daniel Gruner
Professor Michael Kearney
Professor Jennifer Murrow

Preface

This research was conceived and carried out to answer questions about secretive marshbird presence in marshes of the Washington, DC metropolitan area. However, in the process, much additional useful data on urban marshes was gathered. The five main chapters of the dissertation focus on variables, models, and results directly applicable to secretive marshbirds. The first chapter provides an introduction, background information, and literature review on what is known about secretive marshbird species thus far. It also provides justification and context for the work that was undertaken. Chapters two through five provide more specific introductions, methods, results and discussions of original research done to address the objectives described in the introduction. Chapter five discusses broader connections, ties together concepts from each of the three research chapters, and connects these ideas back to the original questions and objectives in the introduction. More detailed information on individual habitat variables, such as plant species and invertebrate families identified, can be found in the appendices.

Dedication

I dedicate this work to my parents, who have supported and encouraged me in my love of animals and the outdoors from an age younger than I can remember.

Acknowledgements

I acknowledge the assistance and guidance of staff members and volunteers from the following natural resource management agencies and organizations: the District Department of the Environment, Fairfax County Parks and Recreation, Fort Belvoir Nature Preserve, Friends of Dyke Marsh, Friends of Huntley Meadows Park, Langston Golf Course, Maryland Division of Natural Resources, National Park Service Capital Parks-East, Northern Virginia Birding Club, Patuxent Research Refuge, Potomac National Wildlife Refuge Complex, Virginia Department of Game and Inland Fisheries, and Virginia State Parks. I also acknowledge the help of the many friends, fellow students, and family members who made themselves available to help with fieldwork when needed.

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Chapter 1: General Introduction

Background and History

The nature of some marshbird species makes them inherently difficult to study. Marshbirds are species that require marshes for some part of their life history, such as nesting, foraging, or raising their young. While the basic ecology of many marshbird species is well documented, very little is known about the more secretive rails and bitterns. These species have inconspicuous or cryptic coloration with infrequent and/or quiet calls. For these reasons, they are not reliably found using large-scale survey efforts such as the North American Breeding Bird Survey (BBS) or the Audubon Society's Christmas Bird Count (CBC) (Conway 2009).

On top of their inconspicuousness, the sheer rarity of many secretive marshbird species adds to the difficulty of study. Although secretive marshbird nests are occasionally found in close proximity to each other (Vesall 1940, Kushlan 1973), none of the secretive marshbird species are considered colonial and they do not live in flocks (Bent 1926). Even when they are present, they occur in very low numbers and densities. For example, although widely distributed across North America, the American bittern is considered rare to uncommon in most of its range (Wiggins 2006). American bitterns are not spotted on most BBS routes because most routes do not include the marsh habitat in which bitterns reside. When examining BBS routes where American bittern are found, fewer than three individual American bitterns are spotted on those routes on average (Wiggins 2006). In one population study at Agassiz National Wildlife Refuge, Minnesota, there were an estimated 384 bitterns in a16,500 ha marsh – an average density of just 0.02 bitterns per hectare (Azure et al 2000).

In a study monitoring restoration in the Eagle Lake Wetland Complex of Northern Iowa with very extensive bird surveys, they estimated 19 Virginia rails and 55 soras in the entire wetland complex, compared to 171 swamp sparrows and 368 yellow-headed blackbirds using the same wetlands, both considered relatively common wetland species (Fletcher and Koford 2003). Even after accounting for low detection rates, Harms and Dinsmore (2012) estimated densities of 0.004 – 0.03 birds/ha for least bittern, 0.02 – 0.10 birds/ha for Virginia rail, and 0.04 – 0.16 birds/ha for sora in freshwater marshes. In a large-scale multi-state study of king rails around the Great Lakes and Upper Mississippi River drainage basin, out of the 264 sites surveyed, eight king rails were detected at five of those sites in 2008, and five king rails were detected at four of those sites in 2009 (Bolenbaugh et al. 2012). Of the more than 4100 BBS routes, king rails were detected at two to eighteen of them per year between 1966 and 2005, with five to 65 total individual king rails counted each year (Cooper 2008). The rice fields of Arkansas and Louisiana are the only parts of their range where king rails are considered common (Meanley 1953, Cooper 2008). King rails occupy these habitats year round and nests can be found in densities of 0.03 – 0.16 nests/ha. There is little information on densities of individual birds in this part of their range, but assuming two adult birds per nest, that is 0.06 – 0.32 birds/ha (Cooper 2008). Yet, even when considered "common" relative to other parts of their range, king rails only occupy 17 – 35% of the available sites (Pierluissi and King 2008, Pickens and King 2012). The secretive nature of these species combined with their sparse population densities results in difficulty studying them and very little information that is known about them.

Secretive marshbird research has changed substantially over time. Early study methods were limited to chance encounters (Mousley 1939, Vesall 1940, Wiens 1966, Johnsguard 1980) nest searches (Walkinshaw 1937, Meanley 1953, Weller 1961), flushing by dogs (Walkinshaw 1940),

or by-products of other wetland activities, such as songbird banding or hunting (Maxfield 1889, Meanley 1956, Webster 1964). Quantitative dietary studies were done by gizzard analyses from necropsied rails (Meanley 1956, Webster 1964, Horak 1970). Tape-recorded callback surveys have been used to detect secretive marshbirds since the 1970s, but methods varied widely. For example, Zimmerman (1984) played five minutes of calls followed by five minutes of silence for each species; Johnson and Dinsmore (1986) used alternating one-minute loops of calls, while Manci and Rusch (1988) played two minutes of silence followed by male and female calls. These studies determined locations and densities of secretive marshbirds, but results were difficult to compare due to the variety of methodologies. It was not known if the populations were actually denser in some places than others or if variations in survey methods were detecting more birds.

Changes in policy and technology allowed advances in marshbird research. After the adoption of the no-net-loss wetlands policy and the resulting wetland restoration projects of the 1990s (DDOE 2006, Krafft et al. 2009), studies began to focus on bird species in natural vs. restored wetlands (Brown and Smith 1998, Fletcher and Koford 2003). These studies found that secretive marshbird abundance increased with wetland restoration efforts. However, these studies still did not address the imperfect detectability of secretive marshbird species. Because of this, few conclusions about habitat quality or predictions of secretive marshbird occupancy could be made. Telemetry technology was used to research survival rates, nest success, migration habits, and habitat selection (Conway et al. 1994, Flores and Eddleman 1995, Haramis and Kearns 2007). However, because of the difficulty in finding and trapping secretive marshbirds, use of this method was not widespread.

Natural History of Target Species

Secretive species that may nest in the Washington, District of Columbia (DC) area are the American bittern (*Botaurus lentiginosus*), least bittern (*Ixobrychus exilis*), king rail (*Rallus elegans*), Virginia rail (*R. limicola*) and sora (*Porzana carolina*) (Fig.1). Most of the basic ecology known about these species comes from the American ornithologist Arthur Cleveland Bent, who authored the comprehensive volume "Life Histories of North American Marshbirds", originally published by the Smithsonian Institution. The ecologies of these species are somewhat similar, but with key variations in behavior and main food sources. The American bittern breeds across Canada and the northern U.S. In the Mid-Atlantic Region, they begin arriving from their wintering grounds in mid March and peak breeding season is mid-April to mid-May (Bent 1926). They lay three to seven eggs and incubate for 24 days. The main food sources for the American bittern are frogs, fish, and large insects, but there are also records of bittern taking mice, snakes, and lizards if available (Bent 1926). In early fall, American bittern migrate to their wintering grounds in the southern U.S., Central America, and the Caribbean (Bent 1926).

The least bittern breeds in the eastern half of the U.S. and in small areas of the coastal southwestern U.S. In the Mid-Atlantic Region, least bittern begin arriving around late April, with peak breeding season from early May to early June (Bent 1926). The nesting behaviors of the least bittern are more flexible than some other secretive marshbirds in that both parents will brood and feed the young, one parent will continue to brood if the other is killed rather than abandon the nest, and renesting or raising multiple broods per season is common. They lay two to six eggs and incubate for 17-20 days from the laying of the first egg (Bent 1926, Weller 1961). Main food sources for the least bittern are small fish, tadpoles, frogs, and insects (Bent 1926). Feeding habits of the least bittern are unique in that it perches over the water on stems and leaves

King rail (*Rallus elegans*)
Photo credit: Chris, Wikimedia Commons

Sora (*Porzana carolina*)
Photo credit: Cephas, Wikimedia Commons

American bittern (*Botaurus lentiginosus*)
Photo credit: fishandgame.idaho.gov

Virginia rail (*R. limicola*)
Photo credit: wildlife.ohiodnr.gov

Least bittern (*Ixobrychus exilis*)
Photo credit: Alan and Elaine Wilson,
Wildlifenorthamerica.com

Figure 1: Secretive marshbird species possibly present in the Washington, DC metro area during the breeding season

of tall emergent vegetation to hunt for prey rather than wading (Weller 1961). Least bittern return to their wintering grounds in Florida, Central America and the Caribbean starting in early fall (Bent 1926).

King rails breed in the Midwestern and southeastern U.S. in freshwater emergent marshes (Bent 1926). They begin arriving in the Mid-Atlantic in mid-April and peak breeding is late April to late May. They lay six to fourteen eggs and incubate for 22 days from the laying of the first egg (Bent 1926, Meanley 1953). Food sources for king rails are mainly crustaceans, aquatic insects and beetles, small fish, and seeds (Bent 1926, Meanley 1956). King rails winter in the Gulf States (where they use rice fields extensively) and southern Atlantic coast (Bent 1926).

Virginia rails breed in the southwestern U.S., northern U.S., and southern Canada (Bent 1926). These birds begin arriving in the Mid-Atlantic States in early-April with peak breeding activity from mid April through late May (Bent 1926). They lay seven to twelve eggs and incubate for about 15 days (Bent 1926). For food, Virginia rails rely mainly on aquatic insects, worms, and larvae, but they may supplement with seeds (Bent 1926, Horal 1970). Wintering grounds for Virginia rails are in the southern U.S. and Central America although there are a few places along the southeastern and southwestern U.S. coasts where these birds will live year-round. Fall migration generally begins in late September or early October (Bent 1926).

Sora breed in the western U.S., northern U.S., and across Canada (Bent 1926). They begin to arrive in the Mid-Atlantic Region in early April with peak breeding activity from late April to late May (Bent 1926). Sora lay six to eighteen eggs ad incubate for fifteen to nineteen days (Bent 1926, Mousley 1937, Walkinshaw 1940, Walkinshaw 1957). During the breeding season, sora depend on insects and larvae as their main food source (Bent 1926). Later in the year sora depend on plant seeds, especially during fall migration (Bent 1926, Webster 1964, Rundle and

Sayre 1983, Haramis and Kearns 2007a). Sora begin to move to their wintering grounds in the southern U.S., Central and South America, and the Caribbean in October (Bent 1926).

The breeding ranges of these species overlap somewhat in the Mid-Atlantic Region. The DC area is on the fringe of the breeding range for the American bittern, Virginia rail, and sora (Bent 1926, Lincoln 1939). Observations of these species during the breeding season are expected to be rare, if present at all. The king rail and least bittern are expected to breed here, as this area is well within their breeding range (Bent 1926). Detection of these birds is expected to be regular.

Conservation Status of Target Species

From state wildlife action plans, it is evident that there are conservation needs for secretive marshbird species. Virginia and Maryland have named the king rail, Virginia rail, least bittern, and American bittern as species of greatest conservation need (VDGIF 2005, MDNR 2005). Virginia specifically identified that gaps in basic knowledge about these species are barriers to effective conservation (VDGIF 2005). In DC, the American bittern, least bittern sora, and Virginia rail have been listed as species of greatest conservation need (DDOE 2006). Knowing which species currently breed in this area, where breeding activity occurs, and what habitat characteristics are associated with marshbird presence will allow these states to update their wildlife action plans and maximize conservation efforts for species in greatest need.

On a larger scale, secretive marshbirds are generally in decline across their range. The American bittern population of North America is declining overall and so little is known about the least bittern that a population trend cannot be identified (Azure et al. 2000, DDOE 2006). From 1966-2004, the only significant trends in American bittern populations identified from Breeding Bird survey data were declines. These declines ranged from 2.3-10.9% (Sauer et al. 2005, Table 1). King rail Breeding Bird Survey counts have been steadily decreasing since 1966

(Sauer et al. 2005, Fig. 2). Although Breeding Bird Survey data is incomplete for secretive species, available data indicate declines in rails and bitterns.

Table 1: American bittern population trends identified from North American Breeding Bird Survey Data, listed by state and region for which American bittern data was available (from Wiggins 2006, data from Sauer et al. 2005). Underlined P-values indicate significant trends. N is the number of BBS routes on which American bitterns were detected.

		1966-2004	
Region	***N***	**Trend**	***P***
South Dakota	16	-4.4	0.28
Nebraska	5	0.5	0.90
Wyoming	6	5.0	0.28
Colorado	4	7.6	0.16
Kansas	2	-1.0	0.77
North Dakota	34	4.1	0.12
Montana	4	-10.0	0.01
Minnesota	43	-7.7	0.00
Alberta	22	-10.9	0.00
Saskatchewan	41	-4.3	0.00
FWS Region 6	72	2.3	0.23
United States	322	0.3	0.80
Canada	291	-2.3	0.02
North America	613	-1.6	0.06

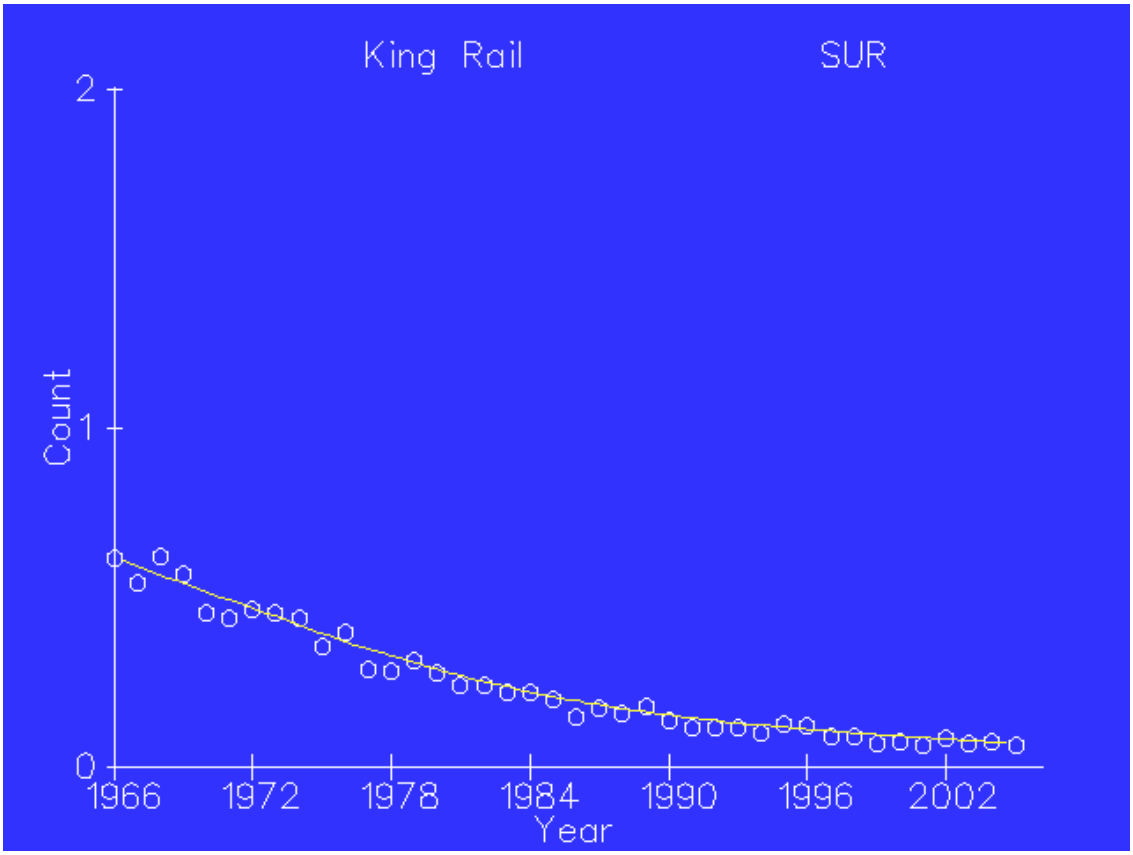

Figure 2: Range-wide king rail counts (birds/route) from North America Breeding Bird Survey routes 1966-2005. Figure from Cooper 2008, data from Sauer et al. 2005.

Declines in secretive marshbird species are also occurring on a global scale. In Africa, Europe, and Asia, the little bittern (*Ixobrychus minutus*), Eurasian bittern (*Botaurus stellaris*), black bittern (*Ixobrychus flavicollis*), and Schrenck's bittern (*Ixobrychus eurhythmus*) are all undergoing range-wide population declines (IUCN 2014). In Great Britain, the Eurasian bittern became locally extinct in the 1880s, but after extensive marsh restoration and fisheries management (the main food source of the Eurasian bittern), the population in Great Britain is now up to a few dozen breeding pairs (Noble et al. 2004, Self 2005). In Australia and New Zealand, the Australasia bittern population has dropped by 77% since monitoring began in the 1970s and they are now endangered in both countries (IUCN 2014, Dept. of the Environment 2015). The New Zealand bittern (*Ixobrychus novaezelandiae*) has been extinct since the 1890s (IUCN 2014). So little information is known about the dwarf bittern of Africa (*Ixobrychus*

sturmii), the pinnated bittern of South America (*Botaurus pinnatus),* and the yellow bittern of southeast Asia (*Ixobrychus sinensis*) that population estimates and trends are unidentifiable. Unfortunately, despite continued habitat loss, because the ranges of these species are large, they remain classified as species of least concern (IUCN 2014). Rail populations are also declining in many areas of the world. The Bogota rail of Columbia (*Rallus semiplumbeus*), and the plain-flanked rail of Venezuela (*Rallus wetmorei*), each an endemic species in their respective areas, are both endangered (IUCN 2014). The austral rail of South America (*Rallus antarcticus*), Mexican highland rail (*Rallus tenuirostris*), and Madagascar rail (*Rallus madagascariensis*) all have declining populations, are endemic to small areas, and are considered near-threatened or vulnerable (IUCN 2014). Of only slightly less concern are the mangrove rail of South America (*Rallus longirostris*), the European water rail (*Rallus aquaticus*), eastern water rail (*Rallus indicus*), and the African rail (*Rallus caerulescens*). These are all classified as species of least concern due to a large range, despite decreasing or even unknown population trends (IUCN 2014). If the decline of these populations becomes more rapid or updated information becomes available, then these species could quickly be elevated to near-threatened or vulnerable. A better understanding of these species and their habitat requirements has the potential to help improve the outlook for secretive marshbird species in other areas of the world.

Current Research

There have been three major recent developments important to research and conservation of secretive marshbirds: first, creation of Rapid Assessment Methods; second, standardization of a reliable survey method; and third, advances in statistical modeling methods appropriate for secretive species. Wetland assessment methods were developed for three main reasons: to address the lack of knowledge of baseline wetland conditions, to prioritize wetland restoration

efforts, and to assess wetland restoration success (Sutula et al. 2007). Many methods were developed by different agencies and organizations, but two have become widespread and popular. One is the Floristic Quality Assessment Index (FQAI). In this method, a coefficient of conservation is assigned to each plant species in every state or region of interest by a panel of expert botanists (Lopez and Fennessy 2002, Chamberlain and Ingram 2012). Higher coefficients are assigned to plant species that are intolerant of disturbance and grow in more pristine sites, while lower coefficients are given to those that will grow in disturbed sites. A higher FQAI would therefore indicate a site that is less disturbed by humans. Another popular wetland assessment method is the California Rapid Assessment Method (CRAM, Stein et al. 2009). This method takes into account biotic, hydrologic, topographic, and landscape factors and is designed to apply across different wetland types (CWMW 2013). Both of these methods are becoming widespread and are now used by many organizations. While they have been shown to correlate with wetland disturbance neither has been evaluated as an indicator of quality marshbird habitat.

The second important recent development in marshbird research is an improved and standardized survey method. A reliable standardized callback survey for secretive marshbirds was not developed until the early 2000s, enabling biologists to achieve much higher detection of secretive species (Conway and Gibbs 2005). This survey method entails playing a set of recorded bird calls in a specific order at specific times of day and year. In this way, results from different studies can be compared and understood easily. However, detection of secretive species is still imperfect and presents data analysis challenges. Traditional statistical modeling methods require data to be divided into sites where birds were present or absent, but with secretive species some proportion of those absences are likely non-detections (Mackenzie et al. 2002). The third major recent development is the creation of new modeling methods to take into account the detection

probability, or the probability that a bird is detected given that it is present at a site (Mackenzie et al. 2002). This can then be used to calculate the likelihood that secretive marshbirds are present at a site based on the habitat characteristics. This is known as occupancy modeling.

These modeling methods have been used in secretive marshbird research in various parts of North America, but not for resident marshbirds of the Mid-Atlantic Region. In coastal Alabama and Mississippi, Rush et al. (2009) found that the best model included cover of salt-tolerant ruches (*Juncus* spp.*)* and amount of water/vegetation interspersion. Rogers et al. (2013) used occupancy modeling to look at the effects of recent controlled marsh burns on king rails (*Rallus elegans*) in North Carolina and southern Virginia. In Louisiana rice fields, occupancy modeling was used to define which characteristics made rice fields preferred habitat for migratory and resident rails (Pierluissi and King 2008; Pickens and King 2012, 2013). In Missouri, occupancy modeling was used to show that tall vegetation, short vegetation, and vegetation/water interspersion were the most important habitat variables for migratory king rails (Darrah and Krementz 2009). However, this modeling method has not been used to find which habitat variables are most important for secretive marshbirds in the Mid-Atlantic Region. This region could potentially play a key role in secretive marshbird distribution due to its central location along the Atlantic flyway, presence of numerous estuarine marshes, and impacts of urbanization (Wilson et al. 2007).

Secretive marshbird habitat requirements vary from region to region and are sometimes unique to a specific area. For example, in the southeastern U.S., the secretive marshbird research revolves around agricultural rice fields (Winstead and King 2006, Darrah and Krementz 2009, 2010, 2011). Factors included in these studies often involve characteristics specific to rice fields, such as drainage ditches, field rotations, neighboring crop type, and flooding regimes. While this

research is important in understanding resident secretive marshbird populations of the southeast as well as some migratory birds that may winter there, the results are unique to that region.

There is regional variation in the available natural habitat types that secretive marshbirds may use. In the prairie pothole region, Naugle et al. (2001) found that local-scale vegetation conditions were more important for rail habitat suitability than landscape-scale factors. In North Dakota, Niemuth and Solberg (2003) showed a positive correlation between number of wetlands in a region and marshbird occurrence. Timmermans et al. (2008) discovered that annual water level fluctuations in Great Lakes coastal marshes affected which wetland bird species were present. However, the wetlands in these studies are depressional or other types of non-tidal marshes. These marshes are very different from the tidal estuarine marshes that dominate the coastal Mid-Atlantic States. There are diverse plant communities and habitats available, and therefore the factors that influence marshbird presence may be different.

Despite its uniqueness relative to other parts of North America, very little research has been done on secretive marshbirds in the Mid-Atlantic States. Most of the research has focused on migratory populations (Haramis and Kearns 2007a,b; Wilson et al. 2007). Sora migrate in large numbers through tidal freshwater marshes of the Chesapeake Bay tributaries and depend on wild rice as a critical food source in the fall (Haramis and Kearns 2007a,b). The only study done during the breeding season was on the Eastern Shore of Maryland in a rural area with restored wetlands (Muir Hotaling et al. 2002). Researchers found that restored wetlands had high marshbird species richness, diversity, and bird abundance. While these results are encouraging, marshbird use of remnant or restored marshes in urban areas may not be the same as in rural areas.

Mid-Atlantic Restoration and Monitoring

Emergent marshes, or wetlands dominated by herbaceous vegetation that emerges from the water, are the critical habitat for all of these species. However, this habitat is relatively rare in DC. The Anacostia and Potomac Rivers border DC. Both rivers have adjacent floodplain areas that historically were marshes. More than 90% of the Anacostia's historic wetlands have been destroyed or altered through direct human activities (DDOE 2006, Krafft et al. 2009), and there has recently been renewed interest in attempting to restore some of this land cover. Over the last 10-15 years, there have been restoration efforts along the Anacostia and Potomac rivers as well as some inland areas to increase emergent marsh and attract marshbird species that historically bred in the area (Krafft et al. 2009). There is evidence that secretive marshbirds will use restored wetlands (Brown and Smith 1998, Muir Hotaling et al. 2002, Peer 2006), but possibly at lower densities or less frequently than natural wetlands (Fletcher and Koford 2003).

In the Mid-Atlantic States, there are some regional marshbird monitoring programs (Fig. 3) that occur during the breeding season. Jug Bay Wetland Sanctuary performs secretive marshbird surveys each spring and has also done research on habitat for migratory sora (Haramis and Kearns 2007 a, b). Scientists at Blackwater National Wildlife Refuge have researched how fire may affect marshbirds, and the Center for Conservation Biology (CCB) in Williamsburg, VA has several marshbird monitoring sites (Fig. 3). However, all of these marshbird monitoring programs are in large marshes in rural areas. These programs are neither near DC, nor do they focus on small, remnant marshes in urban areas. The question remains: Can rails and bitterns use small urban marshes, and if so, what enables them to do so?

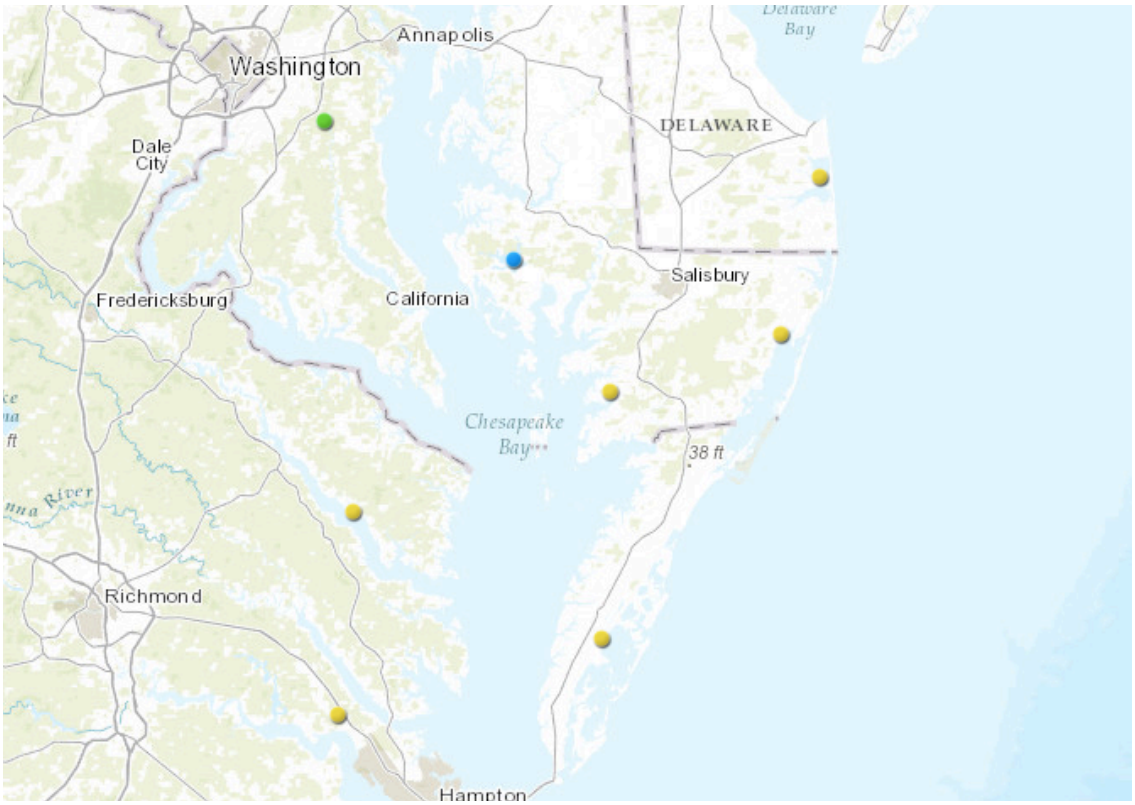

Figure 3: Mid-Atlantic marshbird monitoring programs. Green symbol indicates Jug Bay Wetlands Sanctuary, Blue symbol indicates Blackwater NWR, and yellow symbols indicate Center for Conservation Biology monitoring sites in large, coastal marshes.

Research Objectives

In order to guide habitat restoration efforts in the DC metro area, managers need to know which species are present in the area and their preferred habitat characteristics in this specific area. The objectives of my research project were three-fold: First, to identify the current species and locations of secretive marshbirds present in the DC metro area during the breeding season. This will provide Virginia, Maryland, and the District of Columbia with the updated information they need for their wildlife action plans. This area should be well within the typical breeding range for least bittern and king rail, so I hypothesized that I would find these at several sites. The American bittern, sora, and Virginia rail are regular migrants through the DC area, but it is on the

fringe of their breeding range. I hypothesized that I would not find these species during my surveys.

The second objective was to create an occupancy model to define the habitat characteristics most important for secretive marshbird species. This will guide future restoration and management efforts targeted towards conserving secretive marshbird species. I hypothesized that secretive marshbird presence would be positively correlated with percent cover of tall emergent vegetation, interspersion, percent of time flooded, plant diversity, invertebrate abundance, and buffer width.

The third objective was to explore a correlation between rapid wetland assessment scores and secretive marshbird presence. Because FQAI is strictly based on the conservation coefficients of the plant species identified, I hypothesized that it would be a poor indicator of secretive marshbird presence. I hypothesized that CRAM would be a better indicator of secretive marshbird presence because it takes into account more types of habitat characteristics.

Chapter 2: Secretive Marshbird Breeding Presence in Washington, DC Urban Wetlands

Introduction

Due to inconspicuous coloring and quiet behaviors, secretive marshbirds are not readily detected with common methods, such as the Breeding Bird Survey or Christmas Bird Count (Conway 2009). Secretive marshbirds primarily include rails (*Rallus* spp.) and bitterns (*Ixobrychus* and *Botaurus* spp.). The available evidence indicates that secretive marshbird populations are declining across their ranges proportional to wetland habitat loss (Sauer et al. 2005, Wilson et al. 2007). In order to effectively conserve habitat, it is imperative to study which secretive marshbird species are present in a given area and which wetlands they use. While research on secretive marshbird habitat use has been done in other regions of the U.S., there is limited information for the Mid-Atlantic Region. No research has been done in the Washington, DC metropolitan area despite increasing marsh restoration efforts. Furthermore, most of the research in other regions of the U.S. has been done in more rural areas. As a result, it is largely unknown if secretive marshbirds use urban marshes or what characteristics of urban marshes are associated with secretive marshbird presence.

By 2005, each state in the U.S. had developed a Wildlife Action Plan that identifies species and habitats of greatest conservation need. In Virginia, the black rail (*Laterallis jamaicensis*), American bittern (*Botaurus lentiginosus*), king rail (*Rallus elegans*), least bittern (*Ixobrychus exilis*), yellow rail (*Coturnicops noveboracensis*), Virginia rail (*Rallus limicola*), and clapper rail (*Rallus longirostris*) are all considered species of greatest conservation need (VDGIF 2005). The primary threat to these species is loss of wetland habitat, and Virginia has specifically stated that

gaps in basic knowledge about these species are barriers to effective conservation (VDGIF 2005).

Maryland identified the American bittern, black rail, king rail, and least bittern as species of greatest conservation need (MDNR 2005). Threats to conserving these species in Maryland include lack of scientific understanding of habitat requirements of these species and wetland habitat loss. The Maryland Wildlife Action Plan also identifies the need to inventory, monitor, and research these species in order to understand their habitat needs and critical resources (MDNR 2005).

In Washington, DC, the American bittern, least bittern, sora, and Virginia rail have been listed as species of greatest conservation need (DDOE 2006). The American bittern is thought to occur in low populations with a declining population trend. So little information is available for the other three species that the status and/or trend is entirely unknown (DDOE 2006). The primary challenges in conserving these species within DC are the low quality of marsh habitat and an overabundance of Canada geese (*Branta canadensis*), which decimate marsh vegetation (DDOE 2006, Haramis and Kearns 2007). Based on the wildlife actions plans, it is clear that more information is needed on the status and habitat requirements of secretive marshbird species in order for effective conservation decisions to be made in all three of these jurisdictions.

A reliable detection method was developed in the early 2000s utilizing recorded species-specific calls to elicit responses from secretive marshbirds (Conway et al. 2005). As a result, it is possible to obtain much higher detection rates of these species. The purpose of this study was to employ these survey methods to identify 1) which species were present in the Washington, DC metropolitan area during the breeding season, 2) where these species were residing, and 3) provide a more thorough description of the habitat being used by these species.

Methods

Study Area

The study area was defined as marshes within 50 km of Washington, DC. Potential study sites were identified by searching satellite images of the area for marshes followed by on-the-ground verification that marshes were indeed still present. Sites were eliminated only if they could not be accessed and surveyed within time constraints defined in the North American Marshbird Monitoring Protocol (Conway 2009). This resulted in a final total of 25 separate marshes within 14 different refuges or parks (Table 2).

Table 2: Locations of field sites for secretive marshbird surveys in the Washington, DC metro area

Name of Site	Location	Number of Marshes
Dyke Marsh	Alexandria, VA	1
Featherstone National Wildlife Refuge	Woodbridge, VA	2
Fort Belvoir Nature Preserve	Fort Belvoir, VA	5
Huntley Meadows Park	Alexandria, VA	1
Julie J. Metz wetland mitigation bank	Woodbridge, VA	1
Kenilworth Aquatic Gardens	Washington, DC	2
Kingman Marsh	Washington, DC	1
Leesylvania State Park	Woodbridge, VA	1
Mason Neck National Wildlife Refuge	Lorton, VA	1
Mason Neck State Park	Lorton, VA	1
Mattawoman Creek	Indian Head, MD	1
Occoquan Bay National Wildlife Refuge	Woodbridge, VA	3
Patuxent Research Refuge	Laurel, MD	4
River Terrace Fringe Marsh	Washington, DC	1

All marshes were freshwater marshes with consistent, perennial flooding. This study included both tidal and non-tidal sites (Fig. 4). Marshes were dominated by emergent herbaceous vegetation, such as cattails (*Typha* spp.), sedges (*Carex* spp.), or rushes (*Schoenoplectus* spp.). Marshes were all on public land managed by federal, state, or county agencies. Most of the

marshes were natural, but four along the Anacostia River in Washington, DC had been restored within the last 25 years. In Virginia, one marsh was a constructed wetland mitigation bank and one was a non-tidal marsh that had undergone significant restoration. The marshes on the Patuxent Research Refuge were created as artificial open-water impoundments, which have since developed marsh plant communities.

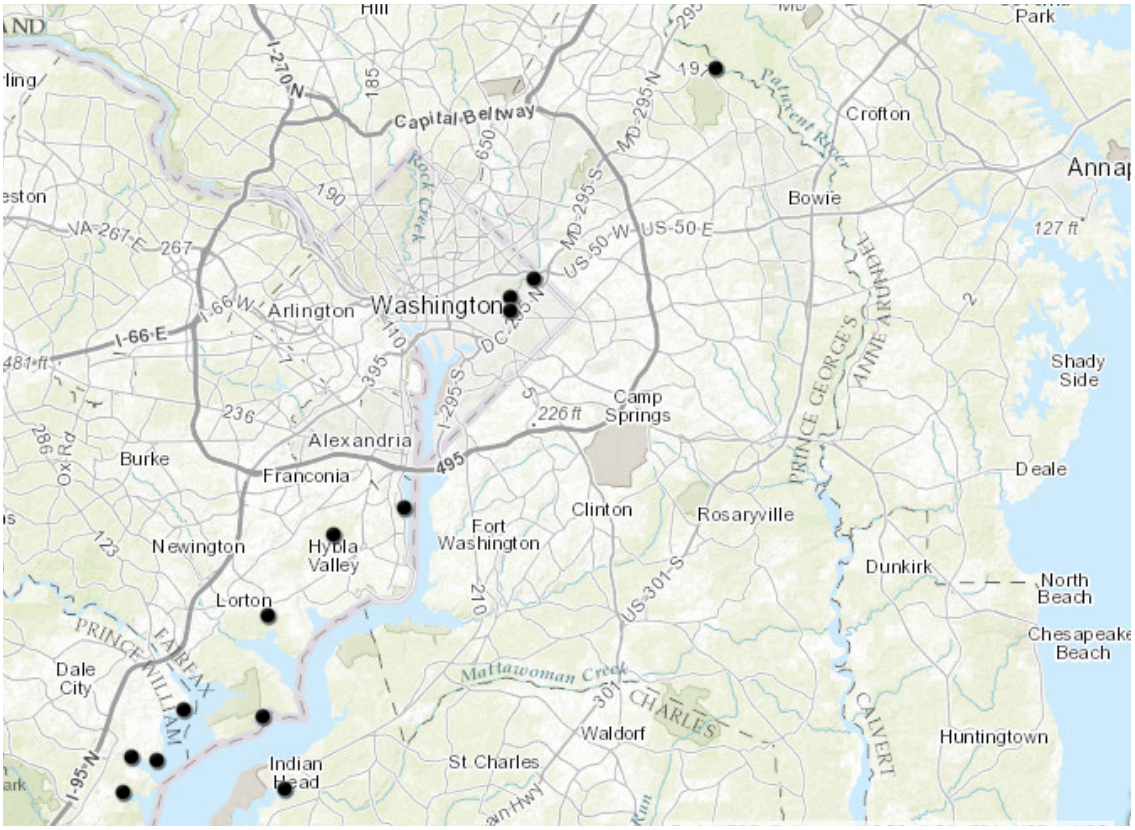

Figure 4: Map of study sites. Note: Some points represent more than one marsh

Marshbird Surveys

Marshbird surveys were conducted following the Standardized North American Marshbird Monitoring Protocol (Conway 2009). Target species in this study were American bittern, least bittern, king rail, Virginia rail, and sora. Points were surveyed either near sunrise or sunset, when

birds are most active. Once a time slot was chosen for a survey, subsequent surveys were performed at the same time and points were surveyed in the same order to avoid confounding factors of time of day.

Marshbird survey points were at least 400 m apart to prevent double counting of birds that may be heard at more than one survey point, as the maximum distance most secretive marshbirds can be heard in an open marsh is approximately 175 m (Conway 2009). Several marshes were large enough to have multiple bird survey locations, resulting in a combined total of 51 bird survey points (see Appendix A for GPS locations). Survey points within a marsh site were chosen randomly, constrained by access to the marsh and the minimum distance of 400m between survey points. Each point was surveyed three times per season within the following time windows: 15-30 April, 1-15 May, and 16-30 May (Conway 2009). This time frame represents peak breeding and nesting activities for the target species and the highest responses to call-broadcast surveys (Conway et al. 2005). This also reflects a time of year after migration is complete for these species, so surveys are targeting breeding resident birds only, not migratory individuals. Surveys were performed in 2013, 2014, and 2015.

At each survey point, the observer listened passively for 5 min, and then played calls of the first target species for 30 sec followed by 30 sec of listening for responses. The 30 sec call/30 sec listening sequence was repeated for each target species. The observer recorded each individual bird detected, its approximate distance from the observer, and during which portion of the survey it was detected. During the last visit of the season, the dominant marsh plant species within a 200 m radius of each survey point was recorded with visually estimated approximate percent cover. Plants species were identified according to Brown and Brown (1984). Native and invasive strains of *Phragmites australis* were distinguished by morphological characteristics described by

Blossey (2002) and Saltonstall et al. (2004). Native *Phragmites* has a shorter, less dense growth form with sparse inflorescences, while invasive *Phragmites* grow in very tall, dense stands with dense inflorescences. Native *Phragmites* has stems that are smooth, shiny, and somewhat flexible with leaf sheaths that readily fall off the stem. Invasive *Phragmites* has stems that are rough, dull, and rigid with leaf sheaths that remain firmly attached to the stem. Additionally, Native *Phragmites* frequently has small black spots from a native fungus that does not colonize the invasive genotype. Cover values for the dominant plant species were averaged over the three years of data collection. Total area of each marsh was measured using Google Earth satellite imagery from 11 April 2015 and the Polygon Measurement Tool (Rypel 2010, Geneva et al. 2013, Barber 2015, Rivera-Milán et al. 2015).

Results

Bird Species Identified

Secretive marshbird species were detected at eighteen of the 51 survey points. Of these, thirteen survey points had least bitterns present, and five had king rails present. Least bitterns were detected at Dyke Marsh, Featherstone National Wildlife Refuge, Huntley Meadows Park, the Julia J. Metz wetland mitigation bank (Neabsco Creek), Mason Neck National Wildlife Refuge, Mattawoman Creek, Occoquan Bay National Wildlife Refuge, and Patuxent Research Refuge (Fig. 5). King rails were detected at Featherstone National Wildlife Refuge, Leesylvania State Park (Powells Creek), and Occoquan Bay National Wildlife Refuge (Fig. 6). Both species were detected in both restored and natural sites. All birds were detected in the high marsh plant community. No American bittern, Virginia rail, or sora were found during bird surveys.

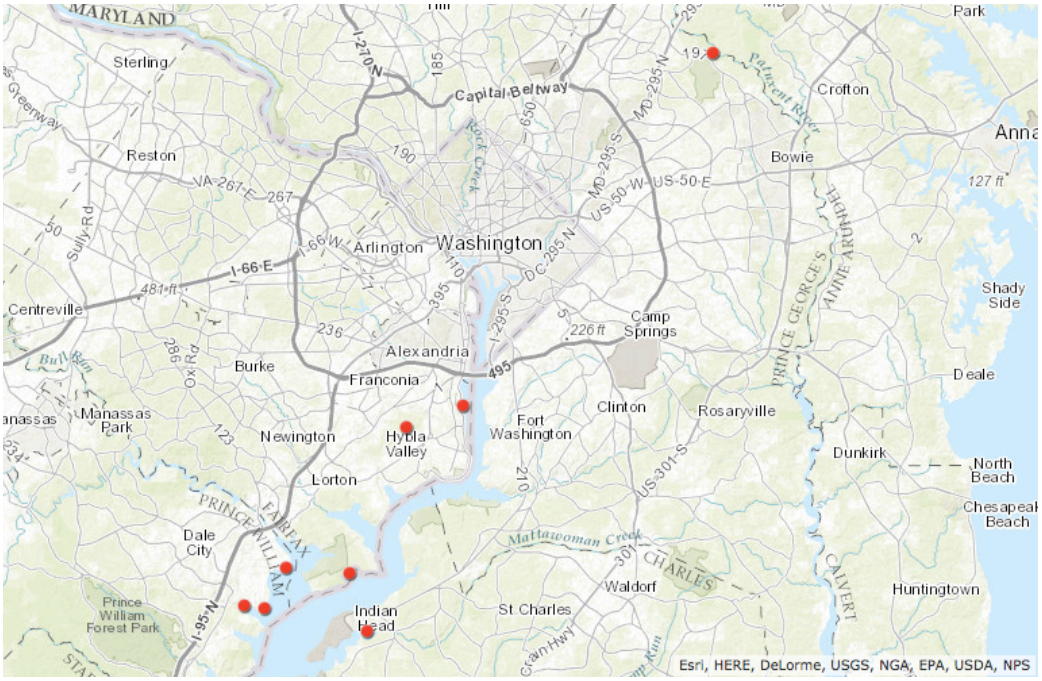

Figure 5: Locations of least bittern detections. Note: Some of the above marshes contain more than one bird survey point where least bitterns were detected.

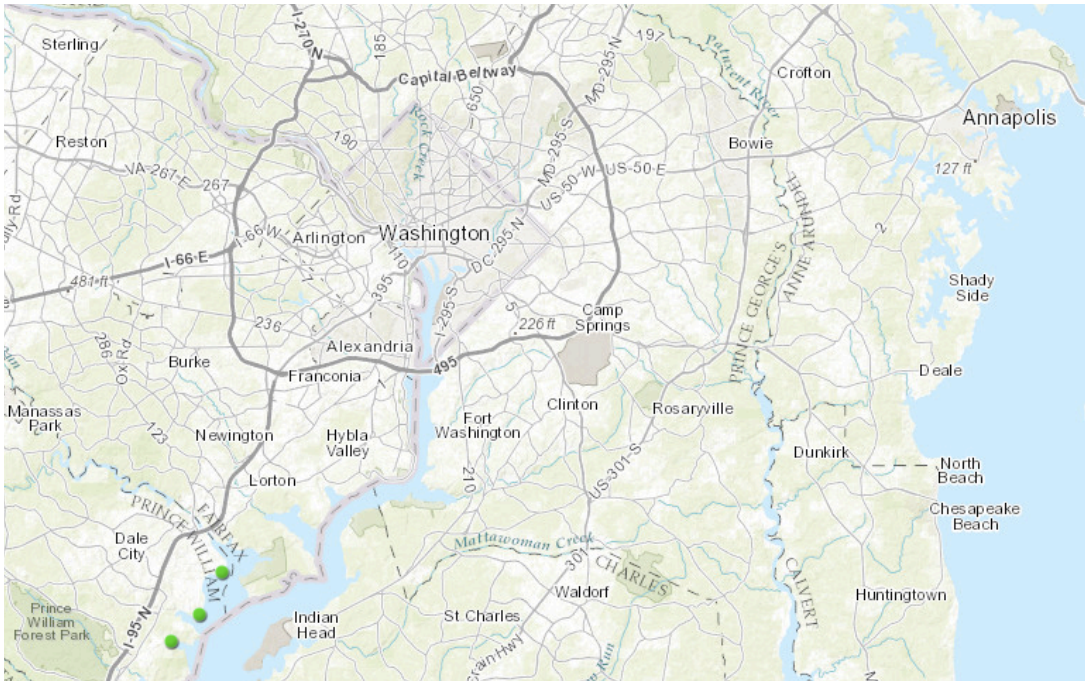

Figure 6: Locations of king rail detections. Note: Some of the above marshes contain more than one bird survey point where king rails were detected.

Marshbird detections varied from year to year (Table 3). Marshbirds were detected at 9 survey points in both 2013 and 2014, despite fewer points being surveyed in 2013. The fewest detections occurred in 2015, with only eight survey points where birds were detected.

Table 3: Number of secretive marshbirds detected in the Washington, DC metro area in 2013, 2014, and 2015, surveyed 3 times per year. NS = not surveyed.

Site Name	**Number of birds detected**									**Species**
	2013			**2014**			**2015**			
Survey number:	**1**	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3**	
Dyke Marsh North	NS	NS	NS	0	0	1	1	0	1	least bittern
Dyke Marsh Lagoon	NS	NS	NS	0	0	0	0	1	0	least bittern
Featherstone NWR South	NS	0	0	0	0	1	0	0	0	least bittern
Featherstone NWR North	NS	0	0	0	0	0	0	1	0	king rail
Huntley Meadows Central Marsh	0	0	0	0	1	0	0	0	0	least bittern
Neabsco Creek North	0	0	0	0	0	0	0	0	1	least bittern
Neabsco Creek Central	0	0	0	0	0	0	0	0	1	least bittern
Neabsco Creek South	0	0	0	0	1	2	0	2	0	least bittern
Mason Neck NWR Central	0	1	0	0	0	2	NS	0	0	least bittern
Mason Neck NWR North	0	1	0	0	2	2	NS	0	0	least bittern
Mattawoman Creek West	NS	NS	0	0	NS	0	0	0	1	least bittern
Powells Creek East	NS	NS	2	NS	0	0	0	NS	NS	king rail
Occoquan Bay Wildlife Drive	0	1	0	0	1	0	NS	0	0	king rail
Occoquan Bay Deephole North	0	1	0	0	1	0	NS	0	0	king rail
Occoquan Bay Deephole South	2	1	1	1	2	0	NS	2	0	king rail
Occoquan Bay Marumsco Creek	0	1	0	0	0	0	NS	0	0	least bittern
Patuxent Refuge Uhler Marsh	NS	1	0	0	0	0	0	0	0	least bittern
Patuxent Refuge Redington	NS	1	0	0	0	0	0	0	0	least bittern
Total # sites marshbirds detected:	**9**			**9**			**8**			

Description of Occupied Habitat

Marshes in this study were 1.2 ha to 274.4 ha. Secretive marshbirds were found in both the largest and smallest marshes surveyed as well as a range of sizes in between. Secretive marshbirds were most often detected in marshes with *Typha* spp. as the dominant vegetation (Table 4). Other dominant vegetation included rushes (*Juncus* spp.), jewelweed (*Impatiens capensis*), the native variety of common reed (*Phragmites australis*), river bulrush (*Scirpus*

fluviatilis), rice cutgrass (*Leersia oryzoides*), the invasive variety of common reed (*Phragmites australis*), common bur-reed (*Sparganium americium*), swamp rose-mallow (*Hibiscus moscheutos*), and smartweeds (*Polygonum* spp.). Mean percent cover of dominant species was 12-44% (Table 5).

Table 4: Secretive marshbird species and numbers of individual birds detected with dominant vegetation type.

Site Name	Species Identified	Dominant Vegetation
Dyke Marsh North	least bittern	*Typha*
Dyke Marsh Lagoon	least bittern	*Typha*
Featherstone NWR South	least bittern	*Phragmites australis* (invasive)
Featherstone NWR North	king rail	*Typha*
Huntley Meadows Central Marsh	least bittern	*Juncus effusus*
Neabsco Creek North	least bittern	*Impatiens capensis*
Neabsco Creek Central	least bittern	*Typha*
Neabsco Creek South	least bittern	*Typha*
Mason Neck NWR Great Marsh Central	least bittern	*Phragmites australis* (native)
Mason Neck NWR Great Marsh North	least bittern	*Phragmites australis* (native)
Mattawoman Creek West	least bittern	*Typha*
Powells Creek East	king rail	*Scirpus*
Occoquan Bay Wildlife Drive	king rail	*Leersia oryzoides*
Occoquan Bay Deephole Point North	king rail	*Sparganium americanum*
Occoquan Bay Deephole Point South	king rail	*Hibiscus moscheutos*
Occoquan Bay Marumsco Creek	least bittern	*Impatiens capensis*
Patuxent Refuge Uhler Marsh	least bittern	*Juncus* spp.
Patuxent Refuge Lake Redington	least bittern	*Polygonum* spp.

Table 5: Dominant vegetation cover at survey points where secretive marshbirds were detected.

Dominant vegetation	Number of Survey Points	Mean cover	Range
Typha spp.	6	28%	13-36%
Juncus spp.	2	30.5%	21-40%
Impatiens capensis	2	36%	29-34%
Phragmites australis (native variety)	2	23.5%	19-28%
Scirpus fluviatilis	1	44%	N/A
Leersia oryzoides	1	23%	N/A
Phragmites australis (invasive variety)	1	25%	N/A
Sparganium americium	1	12%	N/A
Hibiscus moscheutos	1	29%	N/A
Polygonum spp.	1	18%	N/A

Discussion

It is well documented that the study site is well within the breeding range of least bittern and king rail (Bent 1926). Therefore, finding these species at several of the bird survey sites in this study was expected. However, it is interesting to note that there was no overlap between locations where they were detected, i.e. no survey points had both king rails and least bittern. This indicates that although these species inhabit the same general habitat, they may have very different specific habitat requirements. Some differences in basic ecology could account for this. For instance, the least bittern often perches up on stems of larger vegetation to feed (Weller 1961), while the king rail wades in the water or walks along fallen vegetation (Eddleman 1988). More specific habitat characteristics need to be examined in order to define these differences.

The lack of American bittern, Virginia rail, and sora in these sites provides evidence that these species do not breed in the study area or exist in very low numbers and were not detected. American bittern are regularly seen in marshes along the Anacostia River and Patuxent Research Refuge early in the spring, but do not stay through the breeding season (Spencer, personal communication; Rauch, personal communication). These marshes are therefore used for migratory stopover habitat rather than providing habitat for breeding residents.

The year-to-year variation in secretive marshbird detections could mean several things. Secretive marshbirds may have been present, but not detected. Even with a standardized and validated survey method, detection levels can still be low for these species (Conway 2009). This seems the most likely explanation due to the ecology of the species being studied. It could also be that suitable sites were not occupied every year. Secretive generally live in sparse populations (Kushlan 1973, Pierluissi and King 2008, Baschuk et al. 2012), so it could be that in the DC area there were not enough secretive marshbirds to occupy all available quality habitat. Although the

total number of sites where marshbirds were detected decreased from 2013 to 2015, it does not represent a trend of extirpation. While there were some sites where marshbirds were detected in 2013 and not in following years, there were also some sites where birds were not detected in 2013 or 2014, but were detected in 2015. For example, at Neabsco Creek North and Central and at Mattawoman Creek West, no birds were detected in 2013 or 2014, but least bitterns were detected at all of these points in 2015. This does not support a trend of extirpation, but rather lends support to the idea that bitterns were either present and not detected or did not occupy all of these sites every year.

Because secretive marshbirds live in such sparse populations, it is considered normal for these species to only detect on or two individuals at a time, as I did in my surveys. For example, on Breeding Bird Survey routes where American bitterns were detected, fewer than three individual birds were found on average along the entire route (Wiggins 2006). Other studies using methods optimized for secretive marshbird species still find them in very low numbers. For example, Winstead and King (2006) detected one least bittern during 26 of their surveys, and two least bittern during five of their surveys. They never detected more than two least bittern at any survey point. Having secretive marshbirds occur at the low numbers seen in my surveys is normal for these species and is not necessarily indicative of site unsuitability.

Several studies in rural areas have shown that secretive marshbirds will use restored marshes (Brown and Smith 1998, Muir Hotaling et al. 2002, Fletcher and Koford 2003, Peer 2006). The findings of the present study indicate that secretive marshbirds will also use restored marshes in urban areas. While no secretive marshbirds were found in the restored marshes along the Anacostia River, they were found at the restored marsh in Virginia as well as the artificially constructed marshes of the Julia J. Metz wetland bank and Patuxent Research Refuge. This

indicates that secretive marshbirds species will use restored marshes, and that there may be other habitat characteristics that are more important than whether a marsh is natural or restored.

One factor that is of concern to wetland managers is the effects of invasive species. Although invasive species pose a threat to marsh habitat (Wilson et al. 2007), research on marshbirds and invasive plants is scarce and has shown mixed results. Two studies have shown a sharp decrease in marshbird nesting activity with invasive common reed growth (Wells et al. 2008, Meyer et al. 2010). However, Tavernia and Reed (2012) showed that marshbirds will use marshes with invasive purple loosestrife. In agreement with this, the results of my study show that secretive marshbirds will use marshes with stands of invasive common reed as the dominant vegetation. Other invasive species, such as multiflora rose (*Rosa multiflora*), carpgrass (*Arthraxon hispidus*), and Asian dayflower (*Murdannia keisak*) were seen at sites where secretive marshbirds were found. However, none of these invasive species were dominant or occurred in large stands. This finding demonstrates that although invasive species change the vegetation and animal communities, they do not entirely destroy the quality of habitat for secretive marshbirds. Their presence may depend on the specific type and extent of invasion.

Secretive marshbirds were found in a wide range of wetland sizes in this study. In other studies, these species have been found in marshes as small as 0.13 ha (Moore et al. 2009) and as large as 12,000 ha (Manci and Rusch 1988, Evans et al. 1991). The 1.2 ha to 274.4 ha size range of the marshes where birds were detected in this study is well within the expected range. This indicates that secretive marshbirds are able to utilize a variety of marsh sizes, even if they are in an urbanized area.

Other studies of secretive marshbirds indicate that they are most commonly found in cattail-dominated habitats (Post 1998, Bogner and Baldassere 2002, Jobin et al. 2011). However,

it has also been demonstrated that they will use other types of dominant vegetation, such as sedges (Wilson and Long 2011), bulrush (Grove and Henry 1990), or giant cutgrass (*Zizaniopsis miliacea*, Winstead and King 2006). The Mid-Atlantic Region is outside the range of giant cutgrass, so that was not found at any of the study sites. In the Illinois and Mississippi River valley, king rails prefer shorter emergent vegetation and have been found in marshes dominated by rice cutgrass, spikerushes (*Eleocharis* spp.) and smartweeds (Darrah and Krementz 2011). The results of this research corroborate what was found in these other studies. King rails were found in two cattail-dominated marshes, but one of the marshes where king rails were found was dominated by rice cutgrass, similar to Darrah and Krementz (2011). This is the first time that king rails have been documented in marshes with common bur-reed and swamp rose-mallow as the dominant species. These are both tall vegetation species (>1 m tall), but are different than the typical cattail, sedge, and bulrush communities found in other studies.

Least bittern were mostly found in tall vegetation types typical of this species: cattail, common reed, bulrush, and other rushes. This study is the first time least bitterns have been documented with jewelweed or smartweeds as the dominant species. However, in sites where jewelweed or smartweeds were the dominant species rushes and cattails were also common. While most of the secretive marshbirds found in this study were in cattail-dominated marshes, the fact that they were found in several other dominant vegetation types means that these birds are not linked to a specific plant species. There may be other variables, such as plant diversity or vegetation height that are more important than a specific species or community. More in-depth vegetation characteristics need to be examined in order to better understand what habitat characteristics are important for secretive marshbirds.

Although the Standardized Marshbird Monitoring Protocol significantly improves marshbird detection (Conway et al. 2005), detection of secretive species is still a challenge and detection levels are often low. Typical statistical analyses (such as ANOVA or pairwise comparisons) assume that where marshbirds were not detected they are indeed absent. However, in the case of secretive species with low detection, typical statistical analyses are inappropriate because the sites where marshbirds are detected are not true absences (Mackenzie et al. 2002). Comparing percent cover of vegetation in sites where marshbirds were detected versus sites where they were not detected would have limited value because it does not take into account the probability that marshbirds were present and not detected. Although it is useful to have descriptions of where these species might be found, this research should be expanded to include modeling techniques that incorporate detection probabilities. Additionally, more specific habitat variables should be used in order to better understand what habitat characteristics are important for secretive marshbird species.

Chapter 3: Breeding Season Site Occupancy of Secretive Marshbirds in the Washington, DC Metropolitan Area

Introduction

Modern occupancy modeling methods make it possible to conduct more in-depth research on hard-to-detect species. There is strong evidence that secretive marshbird species, such as rails (*Rallus* spp.) and bitterns (*Botaurus* and *Ixobrychus* spp.) are in decline across their respective ranges (Azure et al. 2000, Sauer et al. 2005). Because these birds are secretive and difficult to detect, limited information is available on their ecology and habitat use relative to other wetland birds (Conway et al 2005). Modeling methods based on strict presence/absence, such as logistic regression, are not effective for these species because many of the sites where they are not detected are not true absences (Mackenzie et al. 2002). Because of their extremely secretive nature, the species may be present, but not detected. In these situations, sites are surveyed several times during a period in which site occupancy is closed. In other words, sites are surveyed within a time frame where no individuals are entering or leaving the site (Mackenzie et al. 2002). When a site is surveyed, the target species is recorded as being detected (1) or not detected (0). This series of zeroes and ones for each site is called the detection history (Mackenzie et al. 2002). Occupancy modeling is used to estimate which set of habitat variables best explains species presence (psi), while taking into account the detection probability (p) based on the detection history from repeated surveys using Likelihood Theory (Mackenzie et al. 2002). In likelihood theory, values of psi and p are estimated simultaneously by estimating the most likely combination that results in the observed detection histories. Because probabilities are bounded between 0 and 1 and regression analyses require unbounded variables, psi and p are modeled

using the logit link, or the natural log of the odds. By transforming a probability to odds, that number can be greater than 0, and by taking the natural log of the odds, that number can then be positive or negative (Donovan and Hines 2007). Based on the estimates of the coefficients for each habitat parameter modeled, the solution to the linear equations can then be back-transformed to the probabilities of detection and occupancy. Occupancy modeling is an effective way to determine habitat characteristics important for secretive marshbird presence.

A number of habitat factors may influence secretive marshbird presence. Marsh location is one of these factors. There is evidence that secretive marshbirds may not use habitats in urban areas, especially if there is not a sufficient buffer area between the wetland and human disturbances (Eddleman 1988,Wilson et al. 2007). Wetlands occupied by secretive marshbirds are more likely to be surrounded by natural areas (forest, other wetland types, etc.) than by urbanized areas (Jobin et al. 2011). This indicates that a buffer between the marsh and human disturbance may be important for secretive marshbirds and that marshes with a wider buffer may be preferable.

Because marshbirds rely on marsh plants for nesting materials and on an aquatic food chain (invertebrates, fish, and amphibians), it follows that flooded conditions are important in maintaining critical marshbird habitat (Eddleman 1988). Secretive marshbird presence has been correlated with flooding duration (Pierluissi and King 2008, Timmermans et al. 2008, Moore et al. 2009). Marshes with a higher percent time flooded may provide higher quality habitat and have more secretive marshbirds present.

Marsh plant community characteristics are major factors that should be taken into consideration. Most studies indicate that secretive marshbirds prefer habitats with tall emergent vegetation (>1 m tall), which provides nesting material, cover, and perches for least bittern to

forage (Bogner and Baldassere 2002, Winstead and King 2006, Darrah and Krementz 2009, Jobin et al. 2009, Budd and Krementz 2010, Pickens and King 2013). However, in contrast to this, some studies found that shorter emergent vegetation (<1 m) may be preferred (Darrah and Krementz 2009, 2011). Some marshbirds prefer cattail (*Typha* spp.) to other types of vegetation (Manci and Rusch 1988). Studies seem to be unanimous in finding that secretive marshbirds avoid habitats with woody vegetation encroachment (Meanley 1953, Pierluissi and King 2008, Darrah and Krementz 2009, Pickens and King 2012). The reasons for this are not entirely known, although it has been theorized that larger woody vegetation may provide more perches for predators (Darrah and Krementz 2009, 2010). Higher plant species richness and diversity could provide more types of seeds, shoots, roots, and tubers that may be preferred food items for marshbirds (Perry and Atkinson 2009, Pickens and King 2013). Plants that persist over the winter may provide crucial nesting material for birds arriving in early spring and attempting to build nests (Eddleman 1988). Cover of perennial and annual plants is sometimes associated with the maturity and condition of a wetland ecosystem (Smith and Haukos 2002, Seabloom and Van der Valk 2003). Looking at perennial and annual cover may provide more of an indicator of ecosystem function than taxonomic diversity measures.

Factors related to foraging also influence habitat preferences. Invertebrates are a major part of the diet of most secretive marshbirds (Meanley 1953, Eddleman 1988). Therefore, relative invertebrate abundance may be a good indicator of the suitability of a marsh for secretive marshbirds (Hierl et al. 2007, Baschuk et al. 2012). Invertebrate diversity is a parameter sometimes used to assess the quality of wetlands and may also be an indicator of a high quality habitat for marshbirds (Marchetti et al. 2010). Secretive marshbirds seem to prefer habitats with high open water/vegetation interspersion (Bogner and Baldassere 2002, Winstead and King

2006, Darrah and Krementz 2009, Moore et al 2009, Pickens and King 2012). This edge area created by interspersion may offer easier access to aquatic invertebrates, greater visibility of prey, or more of an influx of food items as water flows through or as tides rise and fall (Budd and Krementz 2010).

Habitat preferences of secretive marshbird species vary with geographic region. For example, in Louisiana, king rails prefer roadside ditches with tall emergent vegetation and flooded rice fields (Meanley 1953, Pierluissi and King 2008), while in the Mississippi and Illinois river valleys they prefer short emergent vegetation cover (Darrah and Krementz 2009). For the least bittern, some researchers found that these birds have a strong preference for cattail-dominated plant communities (Post 1998, Bogner and Baldassere 2002, Jobin et al. 2011), while other studies found that least bittern presence was best explained by hydrology rather than vegetation characteristics (Griffin et al. 2009, Moore et al. 2009, Jobin et al. 2009). With such geographic variation, it is important to have a model specific to the restoration or management region. To date, no habitat analysis of breeding populations of secretive marshbirds has been done in the Washington, DC metropolitan area.

The guiding question for this research was, "What habitat characteristics are important for secretive marshbirds in urban marshes around Washington, DC?" Based on current knowledge of secretive marshbird ecology, it was hypothesized that tall emergent vegetation, marsh area, food abundance, buffer width, percent of time flooded, and plant species diversity are important habitat characteristics for secretive marshbirds in this region. It was also hypothesized that marshbirds are negatively correlated with woody plant encroachment into marshes.

Methods

Study Area

To identify possible study sites, I searched satellite imagery for marshes within a 50 km radius of Washington, DC. These sites were then verified on the ground for presence of marshes. Only sites that could not be accessed and surveyed within time constraints defined in the bird survey method were eliminated (Conway 2006). The final 25 marshes included both tidal and non-tidal marshes ranging in size from 1.2 ha to 274.4 ha, but all were freshwater and perennially flooded (Fig. 7). Dominant vegetation types included cattails (*Typha* spp.), rushes (*Juncus* and *Schoenoplectus* spp.), arrow arum (*Peltandra virginica*), and spatterdock (*Nuphar lutea*).

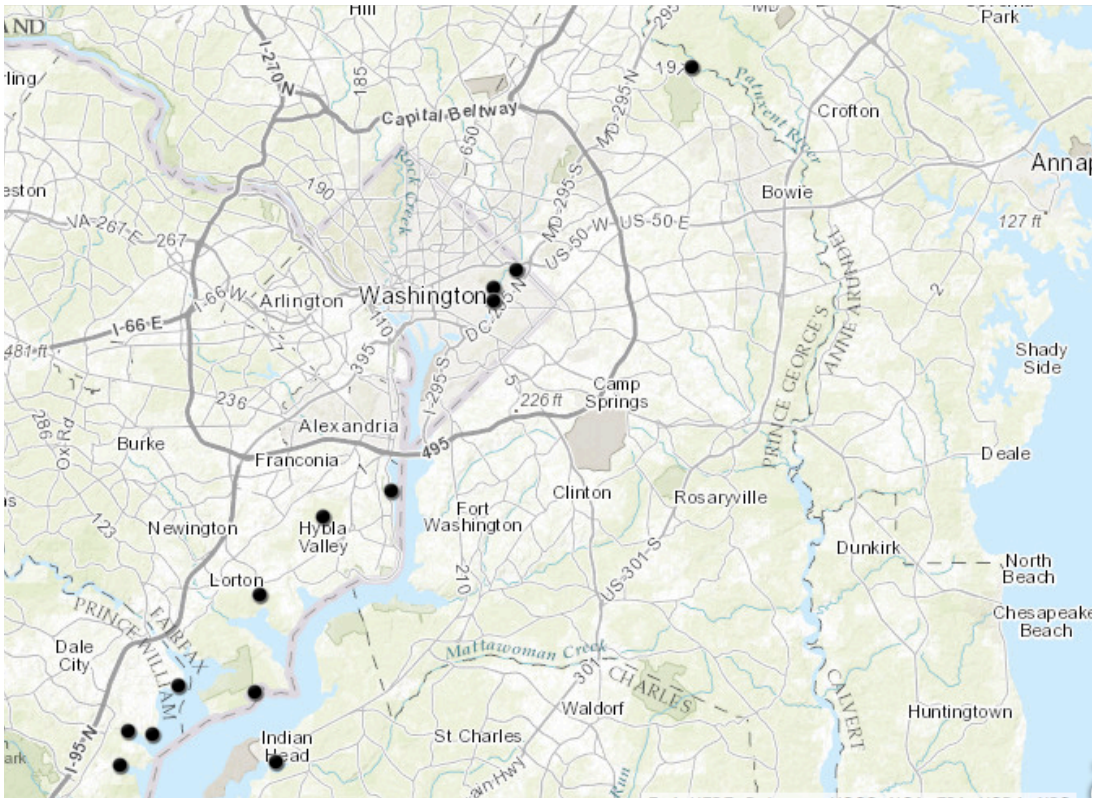

Figure 7: Map of study sites used for occupancy modeling of secretive marshbirds detected in the Washington, DC metro area 2013-2015. Note: Some points represent more than one marsh

Bird Surveys

The target secretive marshbird species for this study were the American bittern (*Botaurus lentiginosus*), king rail (*Rallus elegans*), least bittern (*Ixobrychus exilis*), sora (*Porzana Carolina*), and Virginia rail (*Rallus limicola*). Individual bird survey points were chosen randomly, but constrained by access to the marsh and the necessity of spacing them at least 400 m apart to avoid dual detection of individual birds during the same visit (Conway 2009). This resulted in a total of 51 bird survey locations (see Appendix A for GPS locations). Some of the 25 marshes were large enough to have more than one survey point within a marsh. The number of survey points within a marsh ranged from one to five. Individual bird survey points were still treated as independent points for two reasons: First, because they are far enough apart to have independent detections of birds during each survey and are considered independent in terms of bird detections (Conway 2009); and second, because the 200 m radius area surrounding the survey point is on the home range scale for these birds (Bogner and Baldassere 2002, Pickens and King 2013). This area represents the habitat that marshbirds would be selecting, using, and staying in for the duration of the season. This is the typical approach for these methods with these species, as lumping all survey points together within a larger marsh would obscure habitat differences when one end of a marsh may be very different in marshbird suitability vs. the other end and mask differences in habitat characteristics that are very relevant for these bird species (Winstead and King 2006, Darrah and Krementz 2009, Budd and Krementz 2010, Darrah and Krementz 2010, Pickens and King 2012). Bird surveys were done in accordance with the Standardized North American Marshbird Monitoring Protocol (Conway 2009) in the springs of 2013, 2014, and 2015. This method consists of playing recorded calls around sunrise or sunset when birds are most responsive and recording each individual bird detected. Calls for each of the

five target species were played alternating with 30-second listening periods. Surveys were repeated in each of three survey windows during peak breeding activity: 15-30 April, 1-15 May, and 16-30 May. This time frame also captures breeding birds only, as migration of these species is completed by this time of year. Temperature, wind speed (Beaufort scale), and weather conditions were recorded during each survey (Conway 2009). Repeated surveys resulted in a detection history of detections (1s) and non-detections (0s) for use in occupancy modeling (Mackenzie 2002).

Habitat Variable Data Collection

Occupancy models for secretive marshbirds were created from the following potential habitat variables: total marsh area, interspersion (length of vegetation/water edge), width of buffer around marsh, percent cover of tall emergents, percent woody species intrusion, plant species richness, Simpson Diversity Index of plant species (Peet 1974), proportion of vegetation that persists over the winter, percent cover perennial plants, percent cover annual plants, invertebrate abundance, invertebrate richness, Shannon Diversity Index of invertebrates (Peet 1974), average percent of time flooded, and whether a marsh was restored or natural.

For spatial variables, I used Google Earth Pro with satellite imagery. Because of the lack of information released on Google Earth Pro (GEPro) metadata and proprietary algorithms, there is some doubt about the accuracy of GEPro and its suitability in scientific applications. However, GEPro has been tested for accuracy by comparing it to on the ground measurements and to other GIS remote sensing technologies. Potere (2008) compared the alignment of urban features (such as roads, intersections, and airplane runways) in GEPro satellite imagery to Landsat GeoCover imagery at 436 points in 109 cities worldwide. The GEPro satellite imagery had a horizontal position accuracy of 22.8 m Root Mean Square Error (RMSE). Additionally, imagery of more

developed countries was significantly more accurate than imagery of developing countries ($p<0.01$). Potere concluded that GEPro imagery was suitably accurate for scientific applications that were not on a very fine scale. One drawback to this study is that it only focused on urban areas and gives no information on the accuracy in rural areas.

GEPro has continued to improve since 2008 and other studies have expanded the testing of GEPro imagery. Benker et al. (2011) compared virtual GEPro measurements of distinct geological features in rural southwest Texas to highly precise (<1m) measurements on the ground at 268 points. GPS points from on-the-ground measurements and from GEPro were both imported into ArcMap 9.3.1 for comparison. They found a horizontal position accuracy of 2.64 m RMSE. Benker et al. concluded that GEPro could be used in remote sensing studies. A rebuttal to this paper disputing the formula they used recalculated the horizontal RMSE as 10.52 m rather than 2.64 m (Salinas-Castillo and Paredes-Hernandez 2014). However, Salinas-Castillo and Parades-Hernandez still supported Benker et al.'s conclusions that the accuracy of GEPro has increased compared to 2008 and that it was suitable for remote sensing applications. They recommend that individual users of GEPro imagery evaluate whether it meets the accuracy requirements for their specific studies. In my studies of secretive marshbirds, this level of accuracy should not pose a problem. My research was done in areas with a 200 m radius surrounding the bird survey points, which is at the home range scale for secretive species like the least bittern. I would consider this medium-resolution work that in which a 10.52 m RMSE horizontal accuracy should not pose any problems.

Furthermore, in recent years use of GEPro has become more accepted and used in scientific literature in applications similar to the present study. Roselli and Styles (2012) used Google Earth imagery to measure the area covered by different vegetation types and open water in urban,

semi-urban, and rural wetlands around Bogotá, Columbia. These measurements were used as variables in Principal Component Analysis (PCA) and multiple regression analysis of the abundance of seven endangered wetland bird species. One of these species is the endemic Bogotá rail, a secretive wetland bird with similar ecology to the king rail and Virginia rail that are focal species in the present study. This study was published in the journal Waterbirds, a major source of the current literature eon secretive birds and journal that I would likely be submitting articles to for publication. Pearce and Charlotte (2012) used GEPro to measure area covered by vegetation and water features within 100 m of green roofs in London, England. They then used these as variables in a Generalized Linear Model (GLM) of urban bat habitat. Gomez et al. (2014) used the Measurement Tool in GEPro to measure widths of rivers and included that as one of the variables in Ecological Niche Factor Analysis (ENFA), Maximum Entropy (MaxEnt), and GLM analyses of neotropical river otters (*Lontra Longicaudis*) in Argentina. This article was published in the Journal of Mammalogy, certainly a reputable journal with wide distribution. Rivera-Milán et al. (2015) used the Polygon Measurement Tool in GEPro to measure the area disturbed by forest clearing surrounding survey points for the critically endangered Grenada dove (*Leptotila wellsii*), endemic to the island of Grenada in the Caribbean. This area of forest clearing was used as one of the covariates in a distance-sampling model of dove densities. Selman et al. (2016) studied brown pelican (*Pelecanus occidentalis*) colonies in coastal Louisiana wetlands. They used the Polygon Measurement Tool in GEPro to measure the area of different islands and used this as a variable in a multiple regression model of pelican colony size. These examples represent GEPro used in urban areas and rural areas, in both highly developed and less-developed countries, published in reputable journals, and used as variables in

several types of models. This shows that variables from GEPro can justifiably be used as part of wildlife modeling applications.

Satellite images of each marsh were used to estimate the total area of the marsh, interspersion, buffer width, and percent cover of tall emergent vegetation (Google Earth April 11, 2015 imagery). Marsh area was found by tracing around the edge of the marsh with the Polygon Measurement Tool in GEPro. Interspersion in this study was defined as the total length of vegetation/water edge. It was estimated by tracing all vegetation/water edge within 200 m of a survey point using the Path Measurement Tool in GEPro. I defined the buffer as the area between the marsh/upland edge and the nearest human development. Four buffer width measurements were taken to the north, east, south, and west of each bird survey point and perpendicular to the marsh edge with the Line Measurement Tool in GEPro. The average buffer width used for the model was the mean of these four measurements. Tall emergent vegetation cover was estimated using the Polygon Measurement Tool in GEPro on portions of satellite imagery corresponding to patches of tall emergent vegetation in the field. Percent cover of woody species was estimated visually at each survey point.

To quantify vegetation characteristics, two 100 m^2 vegetation plots were randomly selected in the high and low marsh within the area of sound broadcast around each bird survey point (Fig. 8). This sampling strategy corresponds to the home range scale for secretive marshbirds (Bogner and Baldassere 2002, Pickens and King 2013). In each plot, plants were identified to the species level (Brown and Brown 1984, see Appendix B for plant species identified at each site). Percent cover of each plant species was visually estimated. Species richness was a count of all plant species identified (Peet 1974). The percent cover of each species was averaged across plots to calculate an overall estimate of cover for each species. These values were summed for a total

absolute vegetation cover. Consequently, total cover could be higher than 100% from overlapping plant layers. The proportion (p) of each plant species was calculated as its percent cover over the absolute percent cover for that survey point. The Simpson Diversity Index ($1/\sum_{i=1}^{x} p_i^2$) was calculated for each survey point from the plant species richness and estimated percent cover (Peet 1974). The Simpson Diversity Index was chosen for this variable because it gives more weight to common species, which are the species contributing most to the needs of marshbird (for nesting material, cover, etc.). Proportion of persistent vegetation was the sum of the absolute cover of persistent species over total absolute cover at each point.

Invertebrates were sampled by dip-netting during vegetation sampling. Ideally invertebrates would have been collected at the same time as the bird surveys, but permitting and access conditions prevented this. However, invertebrates were still sampled during a time of year when marshbirds were actively using the marsh and when invertebrates would have been needed as a food resource for fledglings (Bent 1926, Conway 2009). The dip-netting method allows capture of invertebrates that are actively swimming, those settled on the bottom, and those that may be clinging to plant stems, and has been shown to be effective in marshes (Turner and Trexler 1997). A D-ring dip-net was used to take three random 1-m long swipes within the 200 m radius around each survey point. Samples from each survey point were pooled and brought back to the lab for identification to the family level (Thorp and Covich 2001, Voshell 2003, see Appendix C for Invertebrate families found at each site). Numbers of invertebrates in each family were recorded. Invertebrate abundance was the total number of all individuals in all families. Invertebrate richness was the number of families identified. The Shannon Diversity Index was chosen as the index of invertebrate diversity because it gives more weight to rare groups than

other diversity indices (Peet 1974). Rare groups could potentially be indicators of wetland quality or important food items even at low abundance.

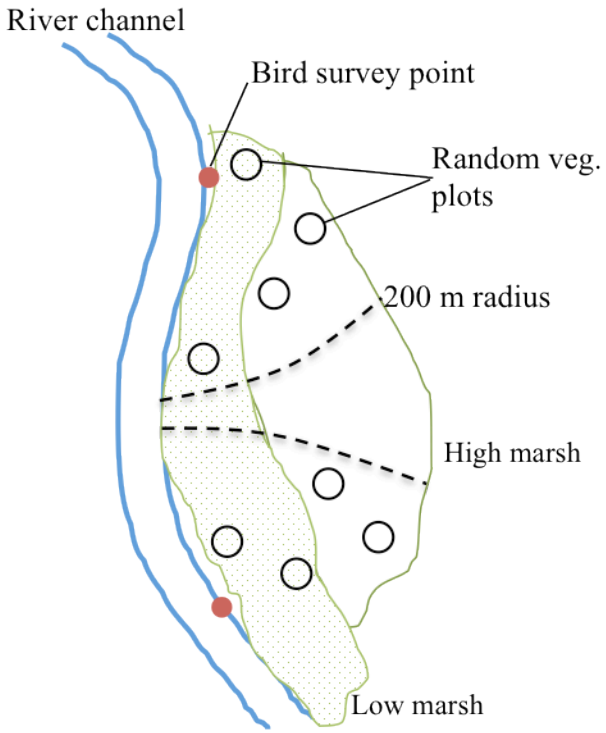

Figure 8: Schematic representation of bird survey points and layout of vegetation plots sampled 2013-2015 for secretive marshbird occupancy modeling in the Washington, DC metro area (not to scale).

To estimate the percent of time flooded in tidal sites, I compared depth measurements from ten test sites at Jug Bay Wetlands Sanctuary (JBWS), Lothian, MD, to water depths recorded by nearby wells installed in marshes at JBWS. I took the difference between the field measurement and the depth at the nearby installed well for the recorded time. To capture water fluctuations during breeding, nesting, and brood rearing, this difference was taken from installed well readings from 15 April to 31 July, 2015 to get depth estimates at study sites for this whole time period. Any study site depth estimates >0 were considered flooded. Percent of time flooded was

calculated as: (number of site depth estimates >0/total number of site depth estimates)*100. Funding and manpower was not available to install wells at all 51 of my bird survey points, so I correlated the percent time flooded at the JBWS test sites to the percent cover of arrow arum (*Peltandra virginica*) at those sites. Arrow arum is a common low marsh plant found at all of my bird survey points and is one of the dominant species of the distinct plant zonation related to the amount of flooding in the low marsh (Mitsch and Gosselink 2000). I performed a simple linear regression in SAS (Proc Reg SAS 9.3, SAS Institute Cary, NC, $y = -0.57x + 62.02$, $R^2 = 0.401$, $p = 0.049$). I then used this linear regression to calculate the estimated percent of time flooded at my 51 bird survey points from the percent cover of arrow arum found at each site. Non-tidal marshes were flooded 100% of the time over this time period, as water levels do not begin to drop substantially until late summer. Consistent flooding in non-tidal sites was confirmed in the field during invertebrate sampling and plant identification.

Occupancy Modeling

Because this study had a small sample size (51 bird survey locations), a model containing all fifteen variables is considered too complex for this dataset. To remove correlated variables, a correlation matrix between all pairs of variables was created in SAS (Proc Corr, SAS 9.3, SAS Institute, Cary, NC). For any pairs of variables with a correlation coefficient greater than 0.7, the one with the stronger biological justification was retained (Lor and Malecki 2006, Hough and Dieter 2009). Variables eliminated via this process were: plant species richness, percent cover perennial plants, percent cover annual plants, invertebrate richness, and Shannon Diversity Index of invertebrates. Total marsh area was also eliminated due to a lack of justification in the literature for any correlation with secretive marshbird presence (see Appendix D for summary of

eliminated variables). Out of twenty studies of secretive marshbird presence, four of them decided to include marsh area as a variable (Benoit and Askins 2002, Lor and Malecki 2006, Rehm and Baldassere 2007, Moore et al. 2009). Of those four, only Lor and Malecki (2006) found any correlation between a marshbird species and marsh area, and they were looking at nests rather than occupancy of adult secretive marshbirds. They found that the odds of finding a Virginia rail nest are 1.1 times larger in a smaller marsh than a larger marsh. They did not find any correlation between marsh area and nests of any other secretive marshbird species studied. All other studies examined either did not include marsh area as a potential variable or looked at adult bird occupancy and found no correlation with marsh area. The final nine habitat variables for occupancy modeling are: interspersion, width of buffer around marsh, percent cover of tall emergents, percent cover woody species, Simpson Diversity Index of plant species, proportion of vegetation that persists over the winter, invertebrate abundance, average percent of time flooded, and whether a marsh was restored or natural (Table 6).

One hurdle to combining variables such as these into occupancy models is the different orders of magnitude between some variables. For example, the Simpson Diversity Index ranges from 1.6 to 10.4, while the interspersion ranges from 76.3 to 1464.9 m. Magnitudinal differences as large as this can make interpretations of model coefficients and relative importance of variables very difficult. Therefore, all variables were converted to Z-scores with the mean at zero and a standard deviation of one using the Standardize Function in Microsoft Excel (Donovan and Hines 2007). Z-scores indicate if a particular value is greater than or less than the mean and by how many standard deviations. For example, a buffer width z-value of -1.5 at a bird survey point indicates that the buffer around that point is 1.5 standard deviations below the mean. By using this scale, variables are standardized for easy comparison and it can readily be seen whether the

values at a particular point are above or below the mean and how strong that deviation is (Donovan and Hines 2007).

These variables were entered into Program Presence (v.10.5 Patuxent Wildlife Research Refuge, Laurel, MD; Mackenzie et al. 2006) along with the bird survey detection histories for model creation (Mackenzie et al. 2002). Program Presence models two probabilities: the probability that a site is occupied and the probability that birds are detected, given that the site is occupied, using maximum likelihood estimates. Both of these probabilities can be modeled as function of habitat covariates using the logit link, or the log of the odds (Mackenzie et al. 2002).

Table 6: Final habitat variables included in secretive marshbird occupancy modeling in the Washington, DC metro area. Units are in parentheses.

Variable Name	**Description**	**Mean (St .Dev.)**	**Range**
Interspersion	Total length of vegetation/water edge (m)	268.8 (210.2)	76.3 – 1464.9
Buffer	Average width of buffer area between marsh and human development (m)	820.2 (698.7)	75.0 – 3051.0
Tall	Percent area covered by vegetation >1 m tall	57.6 (25.0)	10 – 100
Woody	Percent area within marsh with woody vegetation cover	4.9 (6.8)	0 – 25
SimpsonD	Simpson's diversity index, $1/\text{Sum}(p^2)$	5.3 (1.7)	1.6 – 10.4
Persistent	Proportion of vegetation that persist over the winter calculated as the sum of cover of persistent species/total absolute vegetation cover	53.2 (25.5)	5.0 – 131.3
Inverts	Total abundance of individual invertebrates captured at each bird survey point	34.6 (26.4)	6.5 – 131.5
Flooding	Estimated percent of time vegetation around the bird survey point was flooded	82.9 (12.9)	50.2 – 97.9

Restored	Indicates whether a marsh has been restored, has a value of 1 if restored or 0 if natural	N/A	0 -1

To test the influence of covariates on occupancy (psi), a set of *a priori* candidate models was developed based on secretive marshbird species ecology. The initial set of *a priori* models was a set of models for secretive marshbirds in general (Table 7). The nesting habitat model includes tall persistent vegetation that would provide shelter and nesting material in early spring (Eddleman 1988, Manci and Rusch 1988). The marsh condition model includes factors that might be used to evaluate the general condition of a marsh – buffer area between the marsh and human disturbance (Eddleman 1988,Wilson et al. 2007), percent of time the marsh is flooded (Timmermans et al. 2008), and whether it has been restored (Brown and Smith 1998, Muir Hotaling et al. 2002). The vegetation diversity model was based on the idea that diverse vegetation may be higher quality habitat for secretive marshbirds and therefore more likely to be occupied (Perry and Atkinson 2009, Pickens and King 2013). Secretive marshbird species seem to avoid woody vegetation (Meanley 1953, Pierluissi and King 2008, Darrah and Krementz 2009, Pickens and King 2012) and prefer tall vegetation (Bogner and Baldassere 2002, Winstead and King 2006, Darrah and Krementz 2009, Jobin et al. 2009, Budd and Krementz 2010, Pickens and King 2013), so these were both considered important vegetation characteristics to include in this model. The foraging model was created to focus on aspects of marshbird ecology related to feeding habits. Secretive marshbird species feed in flooded areas (Eddleman 1988), often along the vegetation/water edge (Budd and Krementz 2010), and frequently on aquatic invertebrates as their food source (Meanley 1953, Eddleman 1988). The remaining *a priori* models are

combinations of these first four models with some of the factors of feeding habits, vegetation, marsh condition, and nesting habitat thought to be most important for secretive marshbirds.

Table 7: The set of generic *a priori* models for secretive marshbird occupancy (psi) in the DC metro area 2013-2015.

Model
Nesting Habitat Model: psi(persistent+tall)
Marsh Condition Model: psi(buffer+flooding+restored)
Vegetation Diversity Model: psi(SimpsonD+tall+woody)
Foraging Model: psi(flooding+interspersion+inverts)
Nesting Habitat + Diversity Model: psi(persistent+tall+SimpsonD)
Nesting + Marsh Condition Model: psi(persistent+tall+buffer+flooding+restored)
Nesting + Foraging Model: psi(persistent+tall+flooding+interspersion+inverts)
Foraging + Vegetation Model: psi(flooding+interspersion+inverts+tall+persistent+SimpsonD)

Because least bittern and king rails are thought to be the prevalent species in the DC area, additional *a priori* models were developed specific to the ecology of each of these species (Tables 8 and 9). The nesting habitat and marsh condition models are the same for the least bittern, but other models differ. The vegetation *a priori* model focuses more on vegetation structure than diversity due to its habit of perching on vegetation when it forages (Weller 1961, Darrah and Krementz 2010). In the early spring, the vegetation available for perching would be the vegetation that persists over the winter, so proportion of persistent vegetation is also included in the vegetation structure model. The foraging model for the least bittern includes tall vegetation needed for perching, which the generic foraging model does not. The remaining four models are combinations for aspects of the first four thought to be most important for least bittern.

For the king rail, the nesting habitat model, foraging model, and marsh condition model are the same as the generic marshbird models. The vegetation diversity model is different than the generic marshbird model in that it includes the proportion of persistent vegetation in addition to the Simpson Diversity Index, tall vegetation cover, and woody vegetation cover. King rails arrive

earlier and begin nesting earlier in the spring than least bitterns (Bent 1926, Meanley 1953), so they may depend more on persistent vegetation for cover and nesting material when they begin using marshes in the spring.

Table 8: The set of *a priori* models for least bittern occupancy (psi) in the DC metro area from 2013-2015 surveys.

Model
Nesting Habitat Model: psi(persistent+tall)
Foraging Model: psi(tall+inverts+interspersion)
Marsh Condition Model: psi(buffer+flooding+restored)
Vegetation Structure Model: psi(tall+woody+persistent)
Vegetation + Food Source Model: psi(persistent+woody+inerts)
Foraging +Nesting Model: psi(tall+inverts+interspersion+persistent)
Foraging + Vegetation Model: psi(tall+inverts+interspersion+woody+persistent)
Vegetation + Marsh Condition Model: psi(tall+persistent+woody+buffer+flooding+restored)

Table 9: The set of *a priori* models for king rail occupancy (psi) in the DC metro area form 2013-2015 surveys.

Model
Nesting Habitat Model: psi(persistent+tall)
Foraging Model: psi(flooding+interspersion+inverts)
Marsh Condition Model: psi(buffer+flooding+restored)
Vegetation Diversity Model: psi(SimpsonD+tall+woody+persistent)
Nesting + Marsh Condition Model: psi(flooding+restored+tall+persistent)
Foraging + Nesting Model: psi(interspersion+inverts+persistent+tall)
Vegetation + Preferred Food Model: psi(inverts+persistent+SimpsonD)
Foraging + Marsh Condition Model: psi(flooding+interspersion+inverts+buffer+restored)

In addition to the *a priori* models, I ran a regression analysis of all possible models based on bird presence/nondetection in SAS and sorted by Akaike's Information Criterion (AIC) (PROC REG, selection=rsquare AIC, SAS 9.3, SAS Institute, Cary, NC). The AIC is a measure of information lost in the model based on the available data, so a lower AIC indicates a better model (MacKenzie et al. 2006). The top models (lowest AIC) for combined secretive marshbird detections, king rail detections, and least bittern detections were included in Program Presence

occupancy analysis. While regression modeling does not take into account imperfect detection, it still provided insight on important combinations of variables associated with secretive marshbird presence. This approach is not typically taken in habitat or occupancy modeling, but in this case it is justified due to the paucity of specific ecological information available for secretive marshbird species in the Mid-Atlantic Region.

A Comparison Model was also included based on the top model variables from twenty other studies of marshbird habitat occupancy. Out of 20 papers relating to marshbird habitat, nine of them examined variables of water depth or amount of flooding and seven of those nine found a correlation with marshbird use of that habitat. Interspersion was another variable that frequently appeared in the literature. Seven different papers looked at metrics of total vegetation/water edge or vegetation/water edge density, both measurements of interspersion. Five of those seven papers found a positive correlation with marshbird presence. Most papers looked at some aspect(s) of vegetation characteristics with cover of woody vegetation, tall vegetation, and short vegetation being by far the most common. Five papers included cover of woody vegetation as one of the habitat variables of interest, and four of those found a correlation (all negative) with marshbird presence. Six out of 20 papers included cover of tall vegetation and four of those found a positive correlation with marshbird use of that habitat. Six papers also included short vegetation as a variable, but only one found any correlation, so short vegetation was not included in the Comparison Model. The final Comparison Model included percent time flooded, vegetation/water interspersion, percent cover tall vegetation, and percent cover woody vegetation.

Models were constructed as single-season, single-species occupancy models. Single season models were used because there was insufficient data for the extinction and colonization

variables estimated in multi-season models, but Program Presence still treats it as three surveys per season in three separate field seasons (Duncan and Hines 2007). Single species models were used because no bird survey point had more than one species detected, so no interactions between species were suspected. Wind speed, weather conditions, and temperature of each individual survey, and survey window and year were used to model detection probability for the generic marshbird model, king rail model, and least bittern model.

Assessing Model Fit

The models were sorted by AIC adjusted for small sample sizes (AIC_c). The model with the lowest AIC_c was selected as the best-fit model (Burnham and Anderson 2002). The lowest AIC_c may help select the best model, but it does not guarantee that the best model provides a good fit for the data. Goodness-of-fit was tested with 1,000 bootstraps to see how well the selected model fits the data set (Duncan and Hines 2007). This test compares the distribution of the data predicted by the model to the distribution of data from the bootstraps with a Chi-squared test (Mackenzie and Bailey 2004). In this case a lower test statistic and higher p-value indicates a good model fit; i.e., the observed distribution from the bootstraps does not significantly differ from the expected distribution from the model. Top generic marshbird and least bittern models were also tested against independent marshbird survey data from 2011-2015 marshbird surveys at JBWS. The test data can provide an indication of how well the top models predict secretive marshbird detection. Surveys at JBWS were performed three to five times per year at ten bird survey points. There were insufficient detections of king rails at JBWS for testing the king rail model.

Results

Bird Surveys

Secretive marshbirds were detected at 18 of the 51 bird survey points. Least bittern were detected at 13 survey points and king rails were detected at 5 survey points. There were no points where both king rails and least bitterns were detected. No other secretive marshbird species were detected during surveys.

Occupancy modeling

For the generic marshbird model, detection (p) was best modeled by separating the first survey each year from the second and third surveys. Temperature, wind, weather conditions, year, and survey window did not provide good models for detection probability. The top generic marshbird occupancy model was the Vegetation Diversity *a priori* model (Table 10), with a detection probability of 0.05 for the first survey and 0.19 for the second and third surveys. It included Simpson diversity of vegetation, and tall and woody vegetation cover. Coefficients of this model indicate positive correlations between marshbird presence and Simpson's diversity and percent cover tall vegetation, and a negative correlation with woody vegetation cover (Table 13). The top GLM model was the second best model out of the models tested (Table 10) and estimated detection probabilities of 0.06 for the first survey and 0.24 for the second and third surveys. The coefficients of this model show positive correlations with buffer width, tall vegetation cover, and Simpson's diversity, and negative correlations with percent cover woody vegetation and interspersion (Table 13). Models within about two AIC_c of the top model are also considered good models (Burnham and Anderson 2002), so the Nesting Habitat model will also be included in comparisons and discussion. This model includes persistent and tall vegetation cover (Table 10) and showed a small positive correlation with both (Table 13). The Nesting

Habitat model estimated a detection probability of 0.03 for the first survey and 0.23 for the second and third surveys. The Comparison Model of top covariates from other secretive marshbird studies had a delta AIC_c of 4.12 and is therefore considered a poor model relative to some of the *a priori* models and the top GLM model (Table 10).

Table 10: Model results for generic marshbird occupancy (psi) in the Washington, DC metro area, sorted by AIC_c (lower is better)

Model	**AICc**	**Delta AICc**	**AIC weight**	**Model Likelihood**
Vegetation Diversity Model: psi(SimpsonD+tall+woody), p(first survey)	203.72	0	0.4123	1
Top GLM model: psi(interspersion+buffer+tall+woody+SimpsonD), p(first survey)	205.26	1.54	0.1909	0.463
Nesting Habitat Model: psi(persistent+tall),p(first survey)	206.08	2.36	0.1267	0.3073
Nesting Habitat + Diversity Model: psi(persistent+tall+SimpsonD), p(first survey)	206.82	3.1	0.0875	0.2122
Foraging Model: psi(flooding+interspersion+inverts),p(first survey)	207.78	4.06	0.0541	0.1313
Comparison Model: psi(flooding+interspersion+woody+tall),p(first survey)	207.84	4.12	0.0525	0.1275
Marsh Condition Model: psi(buffer+flooding+restored),p(first survey)	207.84	4.12	0.0525	0.1275
Nesting + Marsh Condition Model: psi(persistent+tall+buffer+flooding+restored),p(first survey)	210.4	6.68	0.0146	0.0354
Nesting + Foraging Model: psi(persistent+tall+flooding+interspersion+inverts),p(first survey)	212.33	8.61	0.0056	0.0135
Foraging + Vegetation Model: psi(flooding+interspersion+inverts+tall+persistent+SimpsonD),p(first survey)	213.44	9.72	0.0032	0.0078

For the least bittern, detection was best modeled by using one detection probability for the first survey and a second detection probability for the remaining surveys, as in the generic marshbird models. The top least bittern model was the Top GLM Model (Table 11). Variables included in this model were woody vegetation cover and invertebrate abundance. Variable coefficients from this model indicate that least bitterns were negatively associated with both woody vegetation and invertebrate abundance (Table 13). The top *a priori* model for least bittern was the Vegetation+Food Preference Model, which included persistent vegetation, woody vegetation cover, and invertebrate abundance (Table 11). This model also showed a negative correlation with woody vegetation and invertebrate abundance, with a slight positive correlation

with tall vegetation (Table 13). The detection probability estimated for both these models was very low for the first survey, only 0.03, while the detection probability for the remaining surveys was 0.22. The Comparison Model from variables important in other studies had a delta AIC_c value of 10.5 and was therefore considered very poor, ranking lower than most of the *a priori* models (Table 11).

Table 11: Model results for least bittern occupancy (psi) in the Washington, DC metro area, sorted by AIC_c (lower is better)

Model	**AICc**	**Delta AICc**	**AIC weight**	**Model Likelihood**
Top GLM Model: psi(woody+inverts), p(first survey)	132.36	0	0.7248	1
Vegetation + Food Preferences Model: psi(persistent+woody+inverts),p(first survey)	134.92	2.56	0.2015	0.278
Foraging Model: psi(tall+inverts+interspersion), p(first survey)	139.19	6.83	0.0238	0.0329
Foraging + Vegetation Model: psi(tall+inverts+interspersion+woody+persistent), p(first survey)	139.4	7.04	0.0215	0.0296
Foraging +Nesting Model: psi(tall+inverts+interspersion+persistent), p(first survey)	140.6	8.24	0.0118	0.0162
Vegetation Structure Model: psi(tall+woody+persistent), p(first survey)	140.84	8.48	0.0104	0.0144
Comparison Model: psi(flooding+tall+woody+interspersion), p(first survey)	142.86	10.5	0.0038	0.0052
Nesting Habitat Model: psi(tall+persistent), p(first survey)	145.54	13.18	0.001	0.0014
Vegetation + Marsh Condition Model: psi(tall+persistent+woody+buffer+flooding+restored), p(first survey)	145.71	13.35	0.0009	0.0013
Marsh Condition Model: psi(buffer+flooding+restored), p(first survey)	146.85	14.49	0.0005	0.0007

King rail detection was best modeled as constant across all surveys. The Top GLM Model ranked highest with the lowest AIC_c and estimated a detection probability of 0.27. This model included invertebrate abundance and Simpson's diversity index of vegetation (Table 12). King rail presence was positively correlated with both variables (Table 13). The top *a priori* model for king rail presence was the Vegettion+Preferred Food Model, which incorporated the invertebrate abundance, proportion of persistent vegetation, and Simpson's diversity index of vegetation (Table 12). This model also predicted a detection probability of 0.27. Coefficients in this model indicated a strong positive correlation with Simpson's diversity index of vegetation, a positive

correlation with invertebrate abundance, and a small negative correlation with the proportion of persistent vegetation (Table 13). The Comparison Model with variables important in other marshbird studies had the highest AIC_c and was ranked lowest, or poorest of all models tested.

Table 12: Model results for king rail occupancy (psi) in the Washington, DC metro area, sorted by AIC_c (lower is better)

Model	**AICc**	**Delta AICc**	**AIC weight**	**Model Likelihood**
Top GLM Model: psi(SimpsonD+inverts), p(constant)	68.67	0	0.7488	1
Preferred Food + Vegetation Model: psi(inverts+persistent+SimpsonD), p(constant)	71.05	2.38	0.2278	0.3042
Vegetation Diversity Model: psi(SimpsonD+tall+woody+persistent),p(constant)	78.26	9.59	0.0062	0.0083
Foraging Model: psi(flooding+interspersion+inverts),p(constant)	78.31	9.64	0.006	0.0081
Foraging + Marsh Condition Model: psi(flooding+interspersion+inverts+buffer+restored),p(constant)	78.75	10.08	0.0048	0.0065
Foraging + Nesting Model: psi(interspersion+inverts+persistent+tall),p(constant)	79.4	10.73	0.0035	0.0047
Nesting Material Model: psi(persistent+tall),p(constant)	81.62	12.95	0.0012	0.0015
Nesting + Marsh Condition Model: psi(flooding+restored+tall+persistent),p(constant)	82	13.33	0.001	0.0013
Marsh Condition Model: psi(buffer+flooding+restored),p(constant)	82.83	14.16	0.0006	0.0008
Comparison Model: psi(flooding+interspersion+woody+tall),p(constant)	85.94	17.27	0.0001	0.0002

Table 13: Coefficients and standard errors for variables in top generic marshbird, least bittern and king rail models for the Washington, DC metro area

Model	Variable	Coefficient	SE
Generic marshbirds:			
Vegetation Diversity Model: psi(SimpsonD+tall+woody),p(first survey)	SimpsonD	1.87	2.28
	tall	1.32	1.57
	woody	-1.91	2.46
Top GLM Model:	interspersion	-0.74	0.55
psi(interspersion+buffer+tall+woody+SimpsonD),p(first survey)	buffer	0.71	0.47
	tall	0.90	0.57
	woody	-1.07	0.65
	SimpsonD	1.08	0.65
Nesting Habitat Model: psi(persistent+tall),p(first survey)	persistent	0.34	0.53
	tall	0.41	0.43
Least bittern:			
Top GLM Model: psi(woody+inverts),p(first survey)	woody	-1.40	0.97
	inverts	-2.05	1.09
Vegetation+Food Preferences Model:	persistent	0.10	0.75
psi(persistent+woody+inverts),p(first survey)	woody	-1.44	1.01
	inverts	-2.01	1.11
King rail:	SimpsonD	2.21	0.96
Top GLM Model: psi(SimpsonD+inverts),p(constant)	inverts	1.14	0.57
Preferred Food+Vegetation Model:	inverts	1.17	0.59
psi(inverts+persistent+SimpsonD),p(constant)	persistent	-0.23	0.82
	SimpsonD	2.41	1.22

Coefficients estimated for habitat variables had a wide range. Interpretation of occupancy modeling can be counterintuitive because they do not relate to the occupancy probability directly, but rather through the logit link, and because the variables have been transformed to the standardized z-scale. For example, the coefficient for Simpson's diversity index of vegetation in the generic marshbird Vegetation Diversity model is 1.87. This is the slope of the line for this variable and indicates that for every one-unit increase in Simpson's diversity index, the natural log of the odds of occupancy increases by 1.87. Because the Simpson's diversity index is on the z-scale, a one-unit increase is an increase of one entire standard deviation. If the Simpson's

diversity index were 0.5 (half a standard deviation above the mean), then assuming all other variables remain constant, the logit of the occupancy probability would equal 1.87*0.5, or 0.935, and the occupancy probability would be 0.71. If the Simpson's diversity index were 1.5, then the logit of the occupancy probability would equal 1.87*1.5, or 2.805, and the occupancy probability would be 0.94. So, although 1.87 seems like a small number, in the case of the logit link with a standardized z-scale for the variable, this would be considered a large coefficient that indicates a very strong correlation. Coefficients as small as 0.10, as in the persistent vegetation variable of the least bittern Vegetation+Food Preference model, indicate a weak correlation with occupancy probability. In this case, each one-unit (or one standard deviation) increase in persistent vegetation results in an increase of just 0.10 in the logit of the occupancy probability. This translates to an increase of about 0.03 in the occupancy probability itself.

Assessing Model Fit

For the generic marshbird models, all top models had evidence of lack of model fit with a high Chi-squared statistic, and low p-value. A p-value of 0.05 or lower is considered strong evidence of lack of fit and all generic marshbird models had p-values considerably less than 0.05 (Mackenzie and Bailey 2004). All top generic marshbird models also had high C-hat values. C-hat is an estimate of overdispersion, which can be another indicator of lack of fit (MacKenzie et al. 2006). A C-hat value close to one supports good model fit, while high C-hat values indicate poor fit (MacKenzie et al. 2006). Both top least bittern models had support for goodness of fit with lower Chi-squared statistics, p-values > 0.05, and C-hat values near one (Table 14). The top king rail models both showed similar support for goodness-of-fit (Table 14). Because there were multiple top generic marshbird and least bittern models with similar AIC_c values, I used model averaging to generate occupancy predictions for the JBWS test sites. In model averaging, model

predictions from different models are combined as a weighted average using the AIC_c weights (MacKenzie et al. 2006).

Table 14: Model error and goodness-of-fit test results for top generic marshbird, least bittern and king rail models for the Washington, DC metro area.

Model	***Chi^2 statistic***	***probability of test statistic ≥ observed***	***C-hat***
Generic marshbirds:			
Vegetation Diversity Model: psi(persistent+tall+woody),p(first survey)	14813.73	0.009	10.71
Top GLM Model: psi(interspersion+buffer+tall+woody+SimpsonD),p(first survey)	7533.90	0.020	5.34
Nesting Habitat Model: psi(persistent+tall),p(first survey)	12177.82	0.004	8.95
Least bittern:			
Top GLM Model: psi(woody+inverts),p(first survey)	964.40	0.165	1.37
Vegetation + Food Preferences Model: psi(persistent+woody+inverts),p(first survey)	954.94	0.165	1.12
King rail:			
Top GLM Model: psi(SimpsonD+inverts),p(constant)	1611.31	0.133	1.55
Preferred Food + Vegetation Model: psi(inverts+persistent+SimpsonD),p(constant)	1514.10	0.120	1.69

When predictive ability of the generic marshbird and least bittern occupancy models were tested against independent survey data, the least bittern occupancy models performed somewhat better (Table 15). The generic marshbird models predicted occupancy probabilities of 0.51 – 0.67 for all JBWS test sites, despite a wide range of detections at test sites, ranging from zero to eight detections over five years. The generic occupancy models actually estimated a lower detection probability for the test site at which there were eight detections than the test site at which there were zero detections, so it did not appear to have very good predictive ability at JBWS. The least bittern models did somewhat better, generally predicting higher occupancy probability at sites where there were more marshbird detections and lower occupancy probability at sites where there were fewer marshbird detections. However, at JBWS1 there were zero detections, yet least

bittern models predicted an occupancy probability of 0.78. At JBWS8, there was only one detection over five years, but the least bittern models estimated an occupancy probability of 0.86. The least bittern models overestimated occupancy at these two points.

Table 15: Occupancy probability predictions from model-averaged top generic marshbird and least bittern occupancy models compared with actual 2011-2015 independent marshbird survey data from Jug Bay Wetland Sanctuary, Lothian, MD. "Detections" is the total number of times a marshbird was detected at that survey point from 2011-2015.

Survey point	Detections	Generic marshbird model predictions	Least bittern model prediction
JBWS1	0	0.65	0.78
JBWS2	1	0.63	0.59
JBWS3	1	0.66	0.50
JBWS4	6	0.66	0.76
JBWS5	4	0.66	0.65
JBWS6	7	0.66	0.52
JBWS7	2	0.51	0.19
JBWS8	1	0.67	0.86
JBWS9	5	0.65	0.82
JBWS10	8	0.61	0.85

Discussion

Percent cover of tall vegetation was included in all of the top generic marshbird models. This was somewhat expected, since tall vegetation is important for marshbird nesting and cover (Aniskowicz 1981, Lor and Malecki 2006, Winstead and King 2006). The strength of the correlation varied, however, ranging from 0.41 (relatively weak) to 1.32 (relatively strong). For least bittern, tall vegetation is important as a perch from which to hunt and they seem to prefer habitats with taller plant species (Weller 1961, Darrah and Krementz 2010). Interestingly, tall vegetation did not appear in any of the top king rail models. Rails are more often found in shorter vegetation than bitterns (Darrah and Krementz 2010, 2011; Lor and Malecki 2006), so it could be that tall vegetation is just not an important factor for king rails. The negative correlation with woody vegetation seen in two of the top generic marshbird models was also not surprising based

on previous research showing that secretive marshbird species avoid woody cover (Winstead and King 2006; Darrah and Krementz 2009, 2010). Woody vegetation cover was also a variable in both the top least bittern models. The negative correlation was also consistently strong, ranging from -1.07 to -1.91, indicating that this variable is an important predictor of marshbird occupancy of a habitat. However, the woody vegetation variable was not in either of the top king rail models. This may mean that king rails have a higher tolerance for woody vegetation in a marsh than least bittern, or it could indicate that woody vegetation simply did not have as strong an effect as other variables.

Simpson's diversity index was an important variable in two of the top generic marshbird models and both of the top king rail models. This diversity index takes into account both the number of plant species and the proportion in which they occur. However, because the proportion is squared, it gives more weight to more common species (Peet 1974). Since the more common plant species contribute the bulk of the ecosystem services important for marshbirds, such as nesting material and shelter, it seemed the most appropriate diversity index to use. Furthermore, the positive correlation with Simpson's diversity index was extremely strong for king rails with coefficients of 2.21 and 2.41 in the top models. This means that for every one unit increase in the standardized Simpson's diversity index variable, it more than doubles the logit of the occupancy probability. Such a strong positive correlation with Simpson's diversity index may indicate that secretive marshbirds prefer a plant community that has a few common species, but is not overly dominated by one type of vegetation. In some of the marshes along the Anacostia River, *Typha* spp. occurred in large, dense stands and sometimes covered 100% of sampling plots. Although *Typha* spp. are certainly an important plant for secretive marshbirds, overwhelming dominance may be detrimental to secretive marshbird habitat. In such cases, it

may be beneficial to control such a dominant species and allow co-dominant vegetation to develop. Simpson's diversity index did not appear in either of the top models for least bittern. This could indicate that least bittern tolerate a wider range of vegetation diversity, or that Simpson's diversity index is not the best habitat metric for least bittern.

Invertebrate abundance was not an important factor in the generic marshbird models, but was included in all least bittern and king rail models. Both marshbird species were expected to have a positive correlation with invertebrate abundance, but least bittern exhibited a very strong negative association, with coefficients of -2.01 and -2.05. High invertebrate abundance may have been an indicator of lower fish abundance, which is often a more important food item for bitterns than for king rails (Bent 1926, Baschuk et al. 2012). King rails have a much higher proportion of invertebrates in their diet and have been found to be associated with greater invertebrate abundance than bitterns (Bent 1926, Baschuk et al. 2012). The poor fit of the generic marshbird model may have been due to conflicting ecological needs of least bittern and king rail. This underlies the importance of building a species-specific model, whenever possible, even with somewhat limited detections. Although these species inhabit the same general habitat types (emergent marshes), they clearly have different specific ecological needs.

Other studies (Brown and Smith 1998, Muir Hotaling et al. 2002, Fletcher and Koford 2003, Peer 2006) have shown that secretive marshbirds will utilize restored marshes and that there is no difference between secretive marshbird use of natural and restored marshes. If this is the case, then this variable would not appear in a top model, as restored marshes would not be correlated with marshbird presence. However, this was thought to be an important variable in the present study due to the historical losses of wetlands and the number of restored sites in the Washington, DC area. The results of the present study show that it was not an important variable for secretive

marshbird presence. It did not appear in any of the top models for generic marshbird detection, least bitterns or king rails. This means that secretive marshbirds were not necessarily discriminating between natural and restored marshes and are using both. Marshbirds were neither negatively nor positively correlated with restored marshes. This study provides strong evidence that marsh restoration is an effective conservation tool for secretive marshbird habitat that can provide quality habitat for secretive marshbirds.

In each of the three sets of models in this study, the top GLM model was one of the best models tested. For least bittern and king rails, the top GLM model was the best model, with the top *a priori* model coming very close in rank. Generally it is preferable to have a set of *a priori* models based on species ecology to avoid models that coincidentally explain a model set, but do not have sound ecological foundations (Mackenzie et al. 2006). However, in the case of secretive species in areas where research has not previously been done, the lack of ecological information may make creation of a sound set of *a priori* models difficult. The Top GLM Model may include a combination of variables that is ecologically sound, yet may not have been previously considered. This provides evidence that supports running all possible models when little information is available on the species of interest. However, this should be done in addition to *a priori* models based on what is known about species ecology and careful model assessment to avoid spurious model combinations.

The Comparison Model created from top variables in other studies ranked considerably lower than the top models in all sets of models tested. For the king rail models, the Comparison Model ranked last out of all models tested. When species have a large range, as in the case of king rails and least bittern, there is considerable variation in plant communities and habitat characteristics. Species that have one set of requirements in one region may have a very different set of habitat

preferences in another region. This study shows that information gathered on a species in one location does not necessarily transfer to another location and underscores the importance of having models tailored to the region of interest in order to have effective understanding of species ecology and conservation.

The detection of least bittern at thirteen points and king rails at five points out of 51, with detection probabilities estimated at 0.22 and 0.27 respectively, may seem very low, but are within the normal range for secretive species. In the Illinois and Mississippi River valleys, Darrah and Krementz (2009) had 83 survey points and detected king rails at 12 of them. In a separate study of the same area, Darrah and Krementz (2010) found least bittern at ten of 83 survey locations is 2006, and at ten out of 114 locations in 2007. In the rice-growing agricultural region of Louisiana, Pickens and King (2012) found king rails at 44 of their 155 total survey locations. One exception to this trend is a 2006 study by Winstead and King in two large, actively managed wetlands in rural Tennessee. In these wetlands, least bittern were considered "abundant" in the local area, rather than the typical "rare" designation for this species. They found least bittern at 49 out of the 50 survey points they had at these two wetlands. Even using the North American Standardized Marshbird Monitoring Method optimized for secretive species, studies had estimated detection probabilities at 0.21 – 0.48 for king rails (Darrah and Krementz 2009, Pickens and King 2012), and 0.16 – 0.58 (Bogner and Baldassere 2002, Budd and Krementz 2010, Darrah and Krementz 2010).

While the least bittern models did a slightly better job at predicting marshbird detections at the JBWS test sites than the generic marshbird models, JBWS was not the most ideal place for testing this model. JBWS is along the Patuxent River, which flows southeastward away from the DC metro area directly into the Chesapeake Bay. In contrast, the study sites used to build the

original model were mostly along the Anacostia and Potomac Rivers. While all of these rivers are in the Chesapeake Bay watershed, on a smaller scale they are in separate watersheds until they reach the Chesapeake Bay, so they will be under somewhat different environmental conditions. Additionally, the watershed for the Patuxent River upstream from JBWS is less developed with more rural areas than the areas surrounding the original survey sites. However, there were the same types of marsh species, similar tidal cycles, same marshbird species potentially present, and similar numbers of detections at JBWS as there were at my bird survey points. My sites had zero to six detections over three years of surveys, and the JBWS test sites had zero to eight detections over five years of surveys, so apparent similar levels of marshbird activity. The ideal way to test my model predictions would have been in a nearby heavily urbanized area, such as Baltimore, MD or Richmond, VA, but there is no marshbird presence or survey data available for these areas. Going further away to an area like Virginia Beach, VA where there are some marshbird survey data sets available would have meant comparing the freshwater systems of my sites to salt marsh systems. These would have had entirely different plant and invertebrate communities with some differences in marshbird species present, so would not have been comparable to variables in my models. In short, while testing at JBWS was not ideal, it was the only nearby marshbird survey data set available, had similar types of marshes, the same marshbird species present, and similar numbers of detections, so was considered suitable for testing the model predictions.

There are also some important relevant unknowns and limitations to the data in the present study. Because I focused strictly on urbanized sites, the number of marshes and survey sites available was limited, which limited the numbers of variables that could be included in occupancy models (Duncan and Hines 2007). With more bird survey points, I may have been

able to test more complex occupancy models, but by expanding the survey radius I would have also been including more rural marshes instead of keeping the focus on urban habitats. This may have introduced confounding factors and obscured results specific to marshbird occupancy of urban marshes. Additionally, while occupancy models do give important insights into habitat characteristics associated with marshbird presence, it does not provide estimates of population size or population dynamics. If surveys were continued over many years, it would be possible to identify overall trends of increase or decrease in marshbird occupancy, but still not give actual estimates of population numbers.

This study highlights the importance of having models tailored to the specific species of interest and the local area. Models from other parts of the range or form habitats under different levels of pressure from human development may not correlate marshbird occupancy with relevant habitat variables in another region. Least bittern in the DC metro area were negatively correlated with invertebrate abundance and woody vegetation encroachment, while king rails were positively correlated with invertebrate abundance and Simpson's diversity index of vegetation. These results highlight species-specific ecological differences than can help to guide restoration or conservation efforts targeted towards these species and their habitats.

Chapter 4: Rapid Wetland Assessments as Indicators of Habitat Suitability for Secretive Marshbird Species

Introduction

Relatively little is known about secretive marshbird life histories compared to other bird groups. Because of their inconspicuous nature, secretive marshbirds, such as rails and bitterns, are not well detected by widespread surveys such as the North American Breeding Bird Survey or the Christmas Bird Count (Conway 2006). However, the information that is available on secretive marshbird species indicates that their populations are in decline across their range (Azure et al. 2000, Sauer et al. 2005). There is evidence that declines are primarily linked to wetland habitat loss, and the wetland restoration efforts of the last 20-30 years have led to increases in wetland bird abundance in some areas of North America (Brown and Smith 1998, Muir Hotaling et al. 2002, Niemuth and Solberg 2003, Wilson et al. 2007).

In conjunction with wetland restoration efforts, managers have sought efficient and cost effective methods to assess wetland quality (Lopez and Fennessy 2002, Cohen et al. 2005, Sutula et al. 2006, Fennessy et al. 2007, Stein et al. 2009). These rapid wetland assessment methods are also used to rank wetlands in order to prioritize labor and funding (Spyreas 2014). Two popular methods are the California Rapid Assessment Method (CRAM), and the Floristic Quality Assessment Index (FQAI). CRAM combines buffer and landscape context, hydrology, physical structure (topography and patch richness), and biotic structure metrics into one overall score to indicate wetland quality (Sutula et al. 2006, Stein et al. 2009, CWMW 2013). While CRAM was shown to be effective in assessing general wetland condition (Stein et al. 2009), it has not been tested as a possible indicator of marshbird habitat. Factors that have been positively correlated with secretive marshbirds are water depth, abundance of nearby wetlands, and vegetation

interspersion (Niemuth and Solberg 2003, Lor and Malecki 2006, Darrah and Krementz 2009, Baschuk et al. 2012,). Given this, CRAM seems likely to be a good indicator of marshbird presence because it incorporates such factors.

Alternatively, FQAI focuses only on the plant community, with a score derived from a list of plant species present (Chamberlain and Ingram 2012). It has been shown to be an effective indicator of wetland quality in several regions of the U.S. (Mushet et al. 2002, Cohen et al. 2004, Bourdaghs et al. 2006, Cretini et al. 2012, Spyreas 2014), as well as a tool to monitor wetland restoration success (Lopez and Fennessy 2002). It has not been tested as an indicator of suitable marshbird habitat. It would be helpful to know whether these two common and popular assessment methods, or a combination of these methods, might be good indicators of quality marshbird habitat.

In this research study, I used three years of secretive marshbird survey data from 51 bird survey locations in 25 marshes in the Washington, DC metropolitan area to assess the relationship between high wetland assessment scores and secretive marshbird presence. The goal of this study was to assess whether high scores of either of these wetland assessment methods indicated secretive marshbird presence. In the case of CRAM, I was also interested in determining if high scores in any of the four submetrics were more highly correlated with secretive marshbird presence than the overall CRAM score. I hypothesized that CRAM would be a good indicator of marshbird presence, while FQAI would be a poor indicator because it does not include other habitat factors that may be important for secretive marshbirds, such as flooding regime or wetland abundance in the local landscape. Furthermore, I also hypothesized that the Hydrology and Biotic Structure attributes of CRAM would provide the best indicator of secretive marshbird presence.

Methods

Study Area

I searched satellite imagery for marshes within a 50 km radius of Washington, DC as potential study sites. This was followed by on-the-ground verification that marshes were currently present. Sites that could not be accessed and surveyed within time constraints defined in the bird survey method were eliminated (Conway 2006). All other sites were kept, resulting in a final total of 25 marshes (Fig. 9). All were freshwater and perennially flooded, but included tidal estuarine, non-tidal riverine, and depressional marsh types (CWMW 2013). Dominant vegetation types included cattails (*Typha* spp.), rushes (*Juncus* and *Schoenoplectus* spp.) and spatterdock (*Nuphar lutea*).

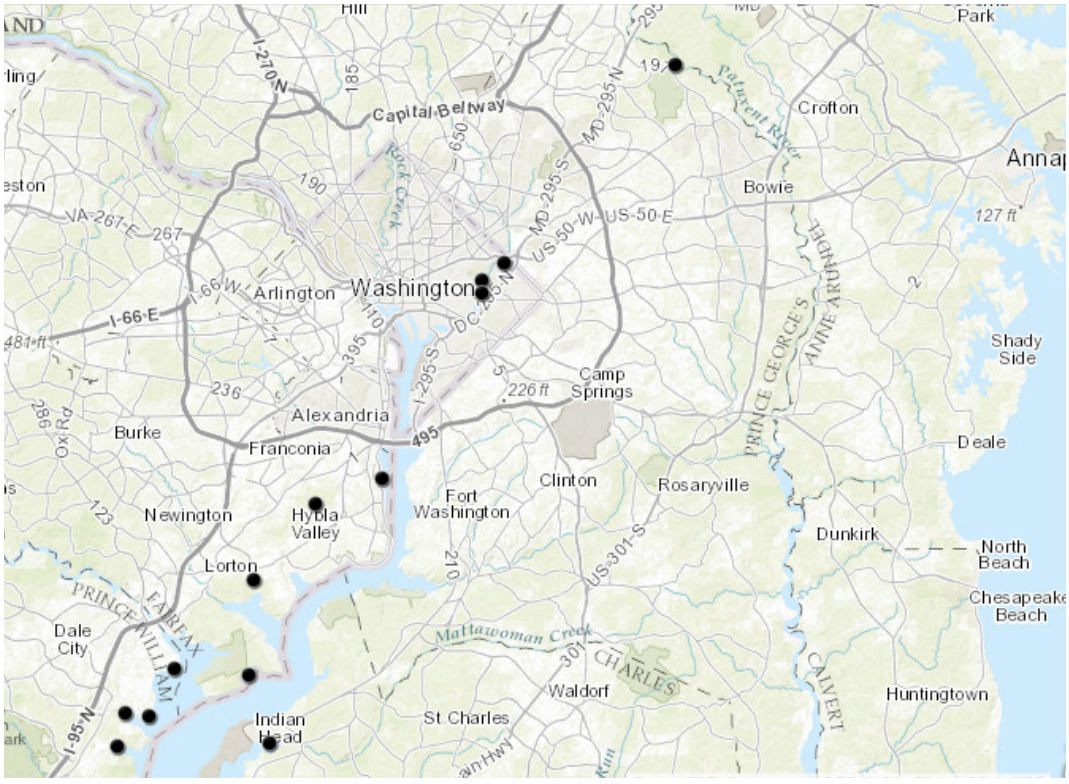

Figure 9: Map of study sites used for occupancy modeling. Note: Some points represent more than one marsh

Marshbird Surveys

The target species in the marshbird surveys were the American bittern (*Botaurus lentiginosus*), king rail (*Rallus elegans*), least bittern (*Ixobrychus exilis*), sora (*Porzana Carolina*), and Virginia rail (*Rallus limicola*). Bird surveys were conducted during the breeding season in 2013, 2014, and 2015, following the Standardized North American Marshbird Monitoring Protocol (Conway 2009). To summarize this method, recorded bird calls are played around sunrise or sunset in each of three survey windows: 15-30 April, 1-15 May, and 16-30 May. This captures both the time of day and time of year when responses to recorded calls are at their peak (Conway 2009). This is also the time of year when migratory birds have already passed through and these surveys detect resident, breeding birds only (Conway 2009). Calls for each of the five target species were played followed by 30-second listening periods between each species. Survey locations are at least 400 m apart to avoid double-counting individual birds (Conway 2009). Some of the 25 marshes were large enough to accommodate more than one bird survey point, resulting in a total of 51 bird survey locations. Repeated surveys allow for the creation of a detection history that can be used in occupancy modeling (Mackenzie et al. 2002).

Rapid Wetland Assessments

CRAM consists of four submetrics: Buffer and Landscape Context, Hydrology, Physical Structure, and Biotic Structure (CWMW 2013). Information for generating scores in each of these metrics was generally obtained from a combination of satellite imagery and field observations. In the Buffer and Landscape Context submetric, the buffer condition is rated on the percent of the marsh that has a buffer, the average buffer width, and the condition of the buffer (disturbed soils, invasive species, human disturbance). The landscape context score increases

with nearby aquatic habitat abundance and continuity of the stream corridor and riparian zone. Google Earth satellite imagery from 11 April 2015 with the Distance Measurement Tool was used to measure average buffer width, percent of marsh with buffer, and percent nearby area with aquatic habitat types. Notes and observations from field visits in 2013, 2014, and 2015 were used to assess buffer condition.

The Hydrology submetric score includes evaluating the water source and any artificial manipulations to water level or water flow, such as dams, dikes, or irrigation withdrawal. To obtain a score for the water source, percent developed or agricultural land within the immediate drainage basin (about 2 km upstream) was visually estimated from satellite imagery (Google Earth, 11 April 2015). Runoff from developed land and irrigated agricultural land provides artificial modifications to natural hydrology and potential for non-point pollution sources (CWMW 2013). Point source discharges and obstructions to water flow, such as dams, dykes, road grades, artificial impoundments or withdrawals from the wetland, or other artificial water control structures were noted during field visits.

There are two parts to the Physical Structure submetric: Structural Patch Richness and Topographic Complexity. Structural Patch Richness is the number of different physical surfaces present in the wetland, such as debris jams, fallen logs, undercut banks, cobbles, plant hummocks, exposed mud, etc. Topographic complexity refers to both macro- and micro-topographic features, such as pools or channels. Wetlands with a greater number of structural patches and more variety in topography receive a higher score. All information for these metrics was obtained from field observations.

The final CRAM submetric is Biotic Structure. This metric includes scores for number of plant layers, number of co-dominant species, percent invasion, horizontal interspersion of plant

communities, and vertical biotic structure (i.e. overlapping plant layers and entrained litter). To obtain data for this submetric, two 100 m^2 vegetation plots were randomly placed in both the high and low marsh within the area of sound broadcast around each bird survey point. In each plot, plants were identified to the species level and percent cover of each plant species was visually estimated. Co-dominant species were defined as those with ≥10% cover in each plant layer present. Interspersion of vegetation types and complexity of vertical structure were noted during field visits.

Scores for each submetric were divided by the maximum possible score for each submetric and converted to a percentage by multiplying by 100 (CWMW 2013). The overall CRAM score was the mean of the four submetric scores.

The plant species list from the vegetation plots described above was used to calculate the FQAI score for each site. In each region of the U.S., a panel of experts assigned conservation coefficients to each plant species (Chamberlain and Ingram 2012). Lower coefficients were assigned to plant species that will readily grow in disturbed areas, while higher coefficients were assigned to plants that are disturbance intolerant, with a maximum coefficient of 10. FQAI is therefore an indicator of how disturbed a site is. The basic formula for FQAI (I) is:

$$I = \bar{C} \times \sqrt{N}$$

where C-bar is the mean conservation coefficient, and N is the number of native species (Chamberlain and Ingram 2012).

However, this basic formula does not take into account non-native species, and it unnecessarily penalizes wetland types with a naturally lower maximum species richness, such as tidal estuarine wetlands vs. depressional wetlands. Therefore, an adjusted FQAI score was developed to account for these two factors. The adjusted FQAI (I') is:

$$I' = \left(\frac{\bar{C}}{10} \frac{\sqrt{N}}{\sqrt{N + A}} \right) \times 100$$

where C-bar is the mean conservation coefficient, N is the number of native species, and A is the number of non-native species (Miller and Wardrop 2006). This adjusted formula accomplishes three things: 1) adds a penalty for presence of non-native species by putting that in the denominator, 2) normalizes Cbar to be out of 10 (the maximum conservation coefficient possible) to eliminate the penalty on communities with a naturally lower maximum species richness, and 3) puts the index on a scale of 100 for easier comparison and interpretation. Because the present study was done in an urban area, there were several invasive species present in most marshes. Additionally, this study included several marsh types. Therefore, the adjusted FQAI was chosen as the appropriate index for this study. Conservation coefficients for each species identified were obtained from the Virginia Department of Environmental Quality (VADEQ, see Appendix E for individual FQAI and CRAM scores for all survey points).

Occupancy Modeling

Secretive marshbirds have low detection even when present at a site. Bird survey detection histories for each site were entered in to Program Presence to allow for modeling detection probability (Patuxent Wildlife Research Refuge, Laurel, MD; Mackenzie et al. 2002, Mackenzie et al. 2006). Overall CRAM scores, individual submetric scores, and FQAI scores were also entered into Program Presence to model marshbird occupancy as a function of the wetland assessments (Table 16). Variables in these models were not standardized because both the FQAI, CRAM scores, and CRAM submetric scores are on the same 0-100 scale and can be directly compared without further transformation or scaling (Duncan and Hines 2007).

Table 16: Summary description of variables included in occupancy modeling

Variable	Summary
FQAI	Floristic Quality Assessment Index
CRAM	California Rapid Assessment Method score
Landscape	Buffer and Landscape Context CRAM submetric score
Hydrology	Hydrology CRAM submetric score
Physical	Physical Structure CRAM submetric score
Biotic	Biotic Structure CRAM submetric score

To test the influence of these different scores on occupancy (psi), a set of *a priori* models was developed (Table 17). These models included CRAM scores, FQAI scores, and sets of CRAM submetric scores thought to be related to secretive marshbird species ecology. Single-season, single-species occupancy models were used in all cases. This setup was considered suitable because there were insufficient detections for the extinction and colonization variables estimated in multi-season models and no bird survey point had more than one species detected. Presence still treats the data as three surveys per year in three separate years. Detection probability was treated as constant across all surveys.

Table 17: The set of *a priori* candidate models for secretive marshbird occupancy related to wetland assessment scores of Washington, DC marshes in 2013-2015.

Model
psi(FQAI)
psi(CRAM)
psi(FQAI+CRAM)
psi(Landscape)
psi(Hydrology)
psi(Physical)
psi(Biotic)
psi(Hydrology+Physical)
psi(Hydrology+Biotic)
psi(Landscape+Biotic)

Models were sorted by Akaike's Information Criterion adjusted for small sample sizes (AIC_c). The model with the lowest AIC_c was considered the best-fit model (Burnham and

Anderson 2002). However, selecting the best model out of all models tested does not guarantee that the model is actually a good fit for the data. Goodness-of-fit was therefore tested with 1,000 bootstraps (Duncan and Hines 2007). This test uses a Chi-squared statistic to compare the distribution of the data predicted by the model to the distribution of bootstrapped data (Mackenzie and Bailey 2004). In this test, a lower test statistic and higher p-value would indicate that the observed distribution from the bootstraps does not significantly differ from the expected distribution from the model. Occupancy predictions of top models and model-averaged predictions of those models were compared to actual survey data from 2011-2015 marshbird surveys at Jug Bay Wetland Sanctuary (JBWS), Lothian, MD to test predictive ability on nearby independent data. Model-averaging is used when there are several models that are close in AIC_c values. In model averaging, occupancy estimates are computer by a weighted average of individual model occupancy estimates based on AIC weight (MacKenzie et al. 2006).

Results

Bird Surveys

Least bitterns were detected at thirteen survey points and king rails were detected at five survey points, for a total of eighteen out of 51 survey points with secretive marshbird detections. There were no points where more than one species were detected. No other secretive marshbird species were detected during surveys.

Occupancy Modeling

The model including FQAI provided the best indicator if secretive marshbird presence (Table 18). The models including both CRAM and FQAI scores, Hydrology, CRAM, Physical, Biotic,

and Landscape variables were within about two AIC_c of the top model, so are also considered good models (Burnham and Anderson 2002).

Table 18: Summary of occupancy modeling results, listed by ascending AIC_c (lower is better)

Model	AICc	deltaAICc	AIC wgt	Model Likelihood
psi(FQAI),p(constant)	210.57	0	0.2752	1
psi(CRAM+FQAI),p(constant)	212.05	1.48	0.1313	0.4771
psi(Hydrology),p(constant)	212.3	1.73	0.1159	0.4211
psi(CRAM),p(constant)	212.65	2.08	0.0973	0.3535
psi(Physical),p(constant)	212.7	2.13	0.0949	0.3447
psi(Biotic),p(constant)	212.81	2.24	0.0898	0.3263
psi(Landscape),p(constant)	212.87	2.3	0.0871	0.3166
psi(Hydrology+Physical),p(constant)	214.42	3.85	0.0401	0.1459
psi(Hydrology+Biotic),p(constant)	214.58	4.01	0.0371	0.1347
psi(Landscape+Biotic),p(constant)	215.16	4.59	0.0277	0.1008

Detection probabilities (p) in the top models ranged from 0.1665 to 0.1704 (Table 19). Coefficients in these models indicated a positive correlation between marshbird presence and FQAI while there was a negative correlation with overall CRAM score and Hydrology, Physical, Biotic, and Landscape submetric scores.

Table 19: Coefficients and detection probabilities with standard errors for variables in top models

Model	Variable	Coefficient	SE	p
psi(FQAI),p(constant)	FQAI	0.107911	0.030977	0.1703
psi(CRAM+FQAI),p(constant)	CRAM	-0.064049	0.014332	0.1665
	FQAI	0.133984	0.034588	
psi(Hydrology),p(constant)	Hydrology	-0.042036	0.011513	0.1689
psi(CRAM),p(constant)	CRAM	-0.027307	0.011218	0.1704
psi(Physical),p(constant)	Physical	-0.013413	0.011806	0.1706
psi(Biotic),p(constant)	Biotic	-0.011853	0.010412	0.171
psi(Landscape),p(constant)	Landscape	-0.000373	0.009975	0.1711

Nearly all models performed poorly in goodness-of-fit tests. The only model with support for a good fit was the model including both FQAI and CRAM scores, with p=0.2547 (Table 20). All other models had substantial evidence for lack of fit with higher Chi-squared statistics and lower

p-values. In addition, all coefficients in all models were very small, indicating weak correlations with marshbird presence.

Table 20: Model error and goodness-of-fit test results for variables in top models

Model	Chi^2 statistic	Probability of test statistic ≥ observed
psi(FQAI),p(constant)	4168.3132	0.037
psi(CRAM+FQAI),p(constant)	4701.8379	0.2547
psi(Hydrology),p(constant)	3762.2856	0.046
psi(CRAM),p(constant)	3880.6757	0.05
psi(Physical),p(constant)	3850.9801	0.0549
psi(Biotic),p(constant)	3798.1198	0.05
psi(Landscape),p(constant)	3740.646	0.0529

When tested for predictive ability on independent marshbird survey data, the model including both FQAI and CRAM scores performed best out of the models tested, i.e. it predicted the highest occupancy at points that had the most actual marshbird detections and low occupancy at points that had fewer marshbird detections (Table 21). The models predictions based on individual submetrics of CRAM were poor. Most of their predicted occupancy probabilities were very similar between points that had very different numbers of detections. The model-averaged estimates of occupancy probability did not do quite as well as the CRAM+FQAI model, but were better than models based on individual submetrics.

Table 21: Occupancy probability predictions from the top occupancy models and model-averaged occupancy estimates compared with independent marshbird survey data from Jug Bay Wetland Sanctuary, Lothian, MD. "Detections" is the total number of times a marshbird was detected at that survey point from 2011-2015.

		Model Predictions							
Point	**Detections**	FQAI	CRAM+FQAI	Hydrology	CRAM	Physical	Biotic	Landscape	Model-averaged estimate
JBWS1	0	0.5276	0.5469	0.5764	0.5081	0.5119	0.4861	0.5044	0.5265
JBWS2	1	0.5273	0.5147	0.4929	0.4945	0.5119	0.4861	0.5044	0.5093
JBWS3	1	0.6079	0.6148	0.4929	0.4945	0.5119	0.4861	0.5044	0.5489
JBWS4	6	0.6684	0.6385	0.4929	0.4706	0.47	0.4861	0.5044	0.564
JBWS5	4	0.7316	0.7628	0.5764	0.4945	0.5119	0.4714	0.5044	0.6183
JBWS6	7	0.5672	0.4369	0.3997	0.4402	0.5119	0.4625	0.5033	0.4896
JBWS7	2	0.5525	0.388	0.3997	0.4267	0.47	0.4714	0.5033	0.4729
JBWS8	1	0.5866	0.5729	0.4929	0.4877	0.5119	0.5128	0.5033	0.538
JBWS9	5	0.584	0.4265	0.3224	0.4267	0.5119	0.4625	0.5033	0.4818
JBWS10	8	0.7267	0.8473	0.3997	0.5557	0.5946	0.5541	0.5044	0.6301

Discussion

Contrary to hypothesized outcomes, secretive marshbird presence had a positive correlation with FQAI, and a negative correlation with CRAM scores. Furthermore, the model including FQAI had a lower AIC_c, meaning it provided a better fit to the marshbird survey data than the model including CRAM scores. However, the model including both FQAI and CRAM scores had a very close AIC_c value, and support for good model fit. While CRAM may incorporate more aspects of marsh condition, our results suggest that CRAM alone does not include an appropriate set of habitat characteristics to indicate quality marshbird habitat. FQAI does not necessarily look at a better set of habitat characteristics than CRAM, but it is a better indicator for secretive marshbird presence. FQAI could potentially be used as a good monitoring or prioritization tool for marshbird conservation. If CRAM is used as the primary monitoring or conservation prioritization tool, then additional research employing species-specific models of habitat variables, such as cover of tall emergent vegetation and abundance of food sources, is

needed to assess habitat suitability for secretive marshbirds. A high CRAM score did not provide an indication of good secretive marshbird habitat in this study.

When tested on marshbird survey data from a nearby marsh, the model that performed best included FQAI+CRAM scores. When using rapid assessment methods across a broader area with marshbird conservation as a goal, it may be most beneficial to use a combination of assessment methods that account for a broader set of habitat characteristics. However, none of the models tested provided a very good fit for the data or exceptional predictive ability. If quality marshbird habitat is one of the conservation goals, I recommend using variables specific to marshbird ecology rather than a general wetland assessment method.

None of the submetrics except hydrology ranked higher than the overall CRAM score based on AIC_c. This serves to re-emphasize the influential role of hydrology on marshbird presence (Eddleman 1988, Darrah and Krementz 2011, Baschuk et al. 2012). However, the negative correlation with hydrology submetric scores was unexpected. Higher hydrology submetric scores indicate little to no alterations to natural flooding regimes (CWMW 2013). Secretive marshbirds nest in emergent herbaceous vegetation over flooded areas (Eddleman 1988, Lor and Malecki 2006). Given this, marshes with natural flooding regimes may have greater water level fluctuations that drown nests. In the Great Lakes region, Timmermans et al. (2008) found that secretive marshbird numbers were more consistent across years in areas with artificially suppressed water level fluctuations, while secretive marshbird numbers fluctuated from year to year with naturally fluctuating water levels. Jobin et al. (2009) also found that wetlands with water levels that are more stable during the breeding season are used more by secretive marshbirds. Artificial modifications to hydrology in this case may provide more stable and predictable water levels and better conditions for breeding secretive marshbirds. However,

impacts on other species must aso be taken into consideration, as changes in hydrology would have impacts on the marsh as a whole.

Chapter 5: Summary and Conclusions

Summary of Findings

There were three main objectives for this research. The first objective was to find out what secretive marshbird species were still breeding in the DC area and which marshes they were using. The second objective was to create an occupancy model that identified what combinations of habitat variables best explained secretive marshbird presence and could be used to guide conservation efforts. The final objective was to assess the ability of two common wetland assessment methods (the Floristic Quality Assessment Index and the California Rapid Assessment Method) to indicate secretive marshbird presence.

After three years of intensive secretive marshbird surveys, I found least bitterns and king rails in several of my field sites during peak breeding times. I did not find any American bittern, sora, or Virginia rail during my surveys. Least bitterns were found in Maryland and Virginia in both tidal and non-tidal marshes. King rails were only found in Virginia in tidal marshes along the Potomac River. Both species were found in a variety of dominant vegetation types and in both restored and natural marshes. This supports previous studies indicating that secretive marshbirds will use restored marshes and that they are not linked to a specific dominant vegetation species. Additionally, marshbirds were found in marshes 1.2 to 274.4 ha in size, showing that they will use a wide range of marsh sizes. If there is a minimum size requirement, it is smaller than 1.2 ha. No secretive marshbirds were found in Washington, DC during bird surveys. However, American bitterns were sighted on Patuxent Research Refuge, Laurel, MD and at Kenilworth Aquatic Gardens, Washington, DC in 2013 and 2014 in early April (before bird surveys began). This suggests that these sites may provide important migratory habitat, but that American bitterns do not breed in the DC area.

The results of the occupancy modeling showed that species-specific models tailored for the region of interest were the best models. The generic marshbird models showed secretive marshbirds positively correlated with tall vegetation, buffer width, and Simpson's diversity index of vegetation; and negatively correlated with interspersion and woody vegetation. Unfortunately, these models did not show good model fit and did not have good predictive ability when tested on nearby independent marshbird survey data. The species-specific least bittern and king rail models showed different patterns. For the least bittern, the top models showed negative correlations with invertebrate abundance and woody vegetation cover and a very slight positive correlation with persistent vegetation. Simpson's diversity index of vegetation, buffer width, and interspersion, and tall vegetation were all factors in the generic marshbird model, but were not important habitat factors in any top models for least bittern. The top king rail models showed positive correlations with invertebrate abundance and Simpson's diversity index and a slight negative correlation with persistent vegetation. Buffer width, woody vegetation, interspersion, and tall vegetation were factors in the generic marshbird models that were not important for king rails. Furthermore, these two species actually had opposite correlation with invertebrate abundance: king rails had a positive correlation, while least bitterns had a negative correlation. The generic marshbird model's poor goodness-of-fit and predictive ability may have been largely due to conflicting habitat needs between king rails and least bitterns. In all sets of models examined, the model based on important habitat variables from other studies in other regions of the U.S. performed considerably worse than most *a priori* models and none came close to the top models, meaning having a model specific to the region of interest is necessary for effective conservation. For secretive species with relatively little information available, assuming that

factors important in other locations apply to a wider geographic range may lead to erroneous conservation and management decisions.

When examining rapid assessment scores, I found that FQAI was more correlated with secretive marshbird presence than CRAM. Although FQAI is solely based on plant species present, it appears that the plant species with higher conservation coefficients also indicate better marshbird habitat. However both of these models individually had some evidence for lack of model fit. The model including both FQAI and CRAM had the most evidence for good model fit. Additionally, the model with the Hydrology submetric of CRAM was ranked as a slightly better model than the overall CRAM score, but had a negative correlation with secretive marshbird presence. Higher Hydrology submetric scores indicate more natural hydrology, lacking artificial modifications such as dykes or dams. A negative correlation with marshbird presence may mean that managed hydrology provides more consistent water levels and better quality habitat for breeding secretive marshbirds. When applied to the JBWS marshbird survey data, a model including both FQAI and CRAM provided the best predictions of secretive marshbird detections. Assessment methods have previously been shown to indicate a marsh that is disturbed or degraded, but scores should be interpreted with caution as indicators of good secretive marshbird habitat.

Implications and Future Directions

I found it very interesting that secretive marshbirds were negatively correlated with both percent time flooded and the hydrology submetric of CRAM. Most other studies showed a positive correlation between marshbirds and flooding or water depth (Lor and Malecki 2006, Timmermans et al. 2008, Jobin et al. 2009, Darrah and Krementz 2011, Baschuk et al. 2012). However, none of these were in tidal wetlands and, thus, were characterized by water level

fluctuations on a different scale. The negative correlation with percent time flooded and the hydrology submetric likely indicates two things: too much flooding could be negatively impacting nesting or breeding of secretive marshbirds (such as drowning nests) or percent time flooded is not the most appropriate habitat variable for modeling secretive marshbird presence. If too much flooding is detrimental to secretive marshbird breeding, then artificial modifications to stabilize hydrology may be helpful for breeding secretive marshbirds. If percent time flooded is not an appropriate variable, then another related variable, such as average water depth while flooded, may provide a better indicator of secretive marshbird presence.

In this study, I only looked at two landscape-scale variables: total marsh area and width of the buffer. Landscape-scale variables in other studies were generally shown to be less important than local scale habitat factors. However, in a heavily urbanized area, such as Washington, DC, where there is more intense land use surrounding the buffer, it may be useful to look at more landscape-scale variables in future research. Rather than just looking at the width of the buffer, it may be interesting to look at the impact of buffer type, i.e. whether it is forest, grassland, open water, a nature park with running trails and heavy foot traffic, etc. There is some debate as to whether or not open water should even be considered a buffer because the impacts of boat traffic and water recreation on marshbirds are entirely unknown (CWMW 2013). Due to the role vocal calls play in bird ecology, noise levels at each site may be something interesting to investigate in the future. However, I did find secretive marshbirds at noisy sites: for example, the Julie J. Metz wetland has a large industrial plant with a noisy water discharge adjacent to the section of the marsh where I found least bitterns. Featherstone National Wildlife Refuge has three marinas and near-constant boat traffic along that section of the Potomac. Leesylvania State Park is extremely popular for recreational kayakers and fisherman (which use both kayaks and

motorized boats), which brings substantial boat traffic into marshes there. In DC the marshes were near busy highways, interstates, and railways and no secretive marshbirds were detected there. It may be that there are certain levels and types of noises that secretive marshbirds tolerate, while others negatively impact their presence.

Another important factor in conserving habitat for secretive marshbirds is climate. Each the three years I did field work was warmer than average for the period of April-July, when there is peak marshbird activity in the marshes. When broken down to monthly averages, every single month was warmer than average except July 2014, which was 0.1 °C cooler than average (Arguez et al. 2010). Warmer temperatures can mean higher water levels due to the thermal expansion of water, besides any additional contributions to water level rise from melting ice. In terms of precipitation, April-July 2013-2015 were all slightly above average, but when looking at months individually, the timing of precipitation may have had an effect on marshbirds. In 2013 and 2015, June had much higher than average precipitation, more than double the average precipitation for that month. More runoff from upland areas also contributes to higher water levels. The month of June could be a critical period when birds are incubating eggs and young are hatching, so high water or flood events during this specific period might destroy nests and negatively impact breeding for those seasons.

Accretion, or the accumulation of sediment and organic matter, builds up marsh soil. Naturally, this process would keep up with any sea level rise and prevent marshes from getting inundated by too much water (Mitsch and Gosselink 2000, Chu-Agor 2011). However, when sea level rise is accelerated by global climate change, accretion rates may not be able to keep up. Marshes that have slow accretion rates in areas with more rapid sea level rise will get flooded out and disappear. Marshes with fast accretion rates can still survive long term. Tidal freshwater

marshes have some of the highest accretion rates of any wetland type, estimated from 1.7 to 15.5 mm/yr along the Potomac River (Glick et al. 2008, Reed et al. 2008) Current sea level rise is estimated at about 3 mm/yr, but is expected to accelerate to 5-10 mm/yr, so despite high accretion rate, some of these marshes will not be able to keep up with that rate of sea level rise (Reed et al. 2008). Another factor that helps determine whether a marsh would persist or disappear is space to move inland. As water levels rise, even if accretion is too slow to keep up, marshes may slowly migrate into newly flooded upland areas (Schile et al. 2014). If, however, a marsh were in a heavily urbanized area directly adjacent to human development or in an area with very step banks, then it would disappear as water levels increase because it has nowhere inland to move.

The Sea Level Rise Affecting Marshes Model (SLAMM) combines factors such as sea level rise, accretion rates, and elevational changes, to predict how sea level rise will affect different marshes (USFWS 2012). According to SLAMM's predictions, over the next 100 years Dyke Marsh Wildlife Preserve would shift to scrub/shrub marsh with more woody encroachment that is less suitable for marshbirds. Featherstone NWR will nearly all disappear as it is next to very high railroad beds that block migration of the marsh into adjacent upland areas. At Leesylvania State Park, Julie J. Metz wetland, Mason neck State Park, Mason Neck NWR, and Occoquan Bay NWR, there is room in low-lying adjacent upland buffer areas for the marsh to migrate inland as water levels rise, so these will still have approximately the same marsh area. Along the Anacostia River, Kenilworth Aquatic Gardens will stay approximately the same, while Kingman and much of the River Terrace Fringe marshes will disappear, with high banks and immediate adjacency to human development. Part of the River Terrace Fringe marsh is along the edge of an island in the middle of the Anacostia River and that part would migrate into upland

areas as water levels rise. The Mattawoman Creek marshes near Indian Head Maryland would almost completely disappear, with very little room to move inland due to local topography and elevational barriers. Conservation and restoration efforts should therefore be focused on marshes that have room to migrate inland or accretion rates than can keep up with sea level rise.

While a better understanding of important habitat factors certainly helps in conserving a species, other conservation issues arise when attempting to conserve species based on habitat factors alone. The occupancy modeling method allows inference on habitat variables correlation with marshbird presence and if repeated over many years, may uncover trends of increase or decrease in marshbird presence. However, this method does not provide information on population dynamics. Unless more in-depth studies on nest success and chick survival are done, there is not a way to know how successful the marshbird occupying these marshes are at contributing to marshbird populations. For example, it is currently unknown whether some of these marshes harbor source or sink populations. Source populations are those in which natality is greater than mortality (a demographic surplus), while sink populations are those in which mortality is great than natality, causing a demographic deficit (Lidicker 1975, Pulliam 1988, Dias 1996). However, without studies on actual population demographics and productivity of the marshbirds, it is unknown whether habitats that are the focus of conservation and restoration efforts will support source or sink populations and how they will contribute to the larger marshbird population. One might assume that a marsh with greater marshbird occupancy is higher quality habitat and might support a source population, but there may be other unknown factors leading to high mortality in that habitat. This becomes what is known as an "ecological trap," where the habitat appears attractive to animals so has high occupancy, but in reality the animals end up dying off rather than reproducing (Dwrnychulk and Boag 1972, Battin 2004). In

the case of secretive marshbirds, an ecological trap could be a habitat with attractive vegetation communities and abundance of food sources, but perhaps high water pollution that interferes with reproductive success, high noise pollution that interferes with mating behavior, or high predation. At this point it is not known if the occupied habitats found in the present study are potential ecological traps.

In order to improve marshes for secretive marshbird species conservation, I recommend management to decrease woody plant incursion, and encourage development of multiple dominant plant species rather than large monocultures. Although marsh size was not in any top models and secretive marshbirds were found in a wide range of marsh sizes, preserving larger marshes may also provide better habitat. King rails and least bitterns were not found together at any individual bird survey site, but they were both found in different areas of Featherstone National Wildlife Refuge and Occoquan Bay National Wildlife Refuge. A larger marsh, or complex of marshes, may provide enough area for development of a variety of habitats to suit both king rails and least bitterns. A smaller marsh would likely only provide habitat for one species or the other. Furthermore, I recommend focusing restoration and conservation efforts on marshes that have higher accretion rates and surrounding adjacent upland areas for inland movement as sea level rises. Marshbird habitat restoration has been shown to be most successful when based on likely future conditions of the marsh rather than on current conditions, even when taking into account the uncertainty of those future conditions (Veloz et al. 2013).

In conclusion, I have discovered which secretive marshbird species are still present in the DC area during the breeding season, where they live, and what habitat characteristics are important for them. I also showed that species specific and region-specific models are crucial to effective conservation or restoration efforts and that wetland assessment scores should be

interpreted with caution when applied to marshbird habitat. This information filled in specific knowledge gaps about these bird species in the Mid-Atlantic region and will be useful for natural resource managers and researchers. This information could have potential international applications if used to guide research and analysis of similar marshbird species on other continents.

Appendix A: GPS Locations of Bird Survey Points

Point Name	Latitude (N)	Longitude (W)
Dyke Marsh North	38.77304	-77.04516
Dyke Marsh Lagoon	38.77131	-77.04856
Dyke Marsh West	38.76872	-77.04909
Dyke Marsh South	38.76535	-77.04773
Featherstone NWR South	38.60391	-77.25815
Featherstone NWR Central	38.60431	-77.25206
Featherstone NWR North	38.60764	-77.24972
Featherstone NWR Creek	38.61178	-77.24808
Fort Belvoir Jack Abbott	38.72953	-77.13350
Fort Belvoir Loop Trail	38.70426	-77.15747
Fort Belvoir Wetlands Trail	38.70004	-77.15922
Fort Belvoir Accotink Marsh	38.69732	-77.16073
Fort Belvoir Accotink Bay	38.69424	-77.15851
Huntley Meadows Marsh Inlet	38.75471	-77.10416
Huntley Meadows Central Marsh	38.75139	-77.10647
Huntley Meadows Creek East	38.74933	-77.11039
Huntley Meadows Creek West	38.74839	-77.11489
Kenilworth Gardens River Trail	38.91158	-76.94664
Kenilworth Gardens Boardwalk	38.91406	-76.94243
Kingman Marsh South	38.90226	-76.96267
Kingman Marsh North	38.90520	-76.96025
Mason Neck NWR Great Marsh West	38.62899	-77.16528
Mason Neck NWR Great Marsh Central	38.63241	-77.16392
Mason Neck NWR Great Marsh North	38.63596	-77.16361
Mason Neck NWR Great Marsh East	38.63675	-77.15907
Mason Neck State Park East	38.65082	-77.17951
Mason Neck State Park Central	38.64951	-77.18543
Mason Neck State Park West	38.64930	-77.19086
Mattawoman Creek West	38.58955	-77.14654
Mattawoman Creek Island	38.58943	-77.14208
Mattawoman Creek Central	38.59243	-77.14040
Mattawoman Creek North	38.59618	-77.13607
Mattawoman Creek East	38.59195	-77.13012
Neabsco Creek West	38.61119	-77.28010
Neabsco Creek North	38.61157	-77.27496
Neabsco Creek Central	38.60983	-77.27070
Neabsco Creek South	38.60805	-77.26796
Occoquan Bay Wildlife Drive	38.64719	-77.23228
Occoquan Bay Deephole Point North	38.64380	-77.22592

Occoquan Bay Deephole Point South	38.64036	-77.22804
Occoquan Bay Marumsco Creek	38.64144	-77.24063
Patuxent Refuge Rogue Harbor	39.08529	-76.73499
Patuxent Refuge Uhler Marsh	39.05693	-76.81317
Patuxent Refuge Knowles North	39.04720	-76.79473
Patuxent Refuge Knowles South	39.04333	-76.79428
Patuxent Refuge Lake Redington	39.02280	-76.80373
Powells Creek East	38.58948	-77.28181
Powells Creek Central	38.58723	-77.27820
Powells Creek South	38.58655	-77.27378
Powells Creek West	38.58558	-77.26955
River Terrace Fringe	38.89350	-76.96136

Appendix B: Plant Species Identified at Bird Survey Points

Site Name	Plant Species Identified
Dyke Marsh North	*Typha angustifolia*
	Mikania scandens
	Apios americana
	Impatiens capensis
	Peltandra virginica
	Nuphar lutea
	Sagittaria latifolia
	Calystegia sepium
	Vitis riparia
	Bidens spp
	Leersia oryzoides
	Bolboschoenus fluviatilis
	Hibiscus moscheutos
	Pontederia cordata
	Amaranthus cannabinus
	Zizania aquatica
	Acorus calamus
	Boehmeria cylindrica
	Schoenoplectus tabernaemontani
	Commelina virginica
Dyke Marsh Lagoon	*Onoclea sensibilis*
	Typha angustifolia
	Peltandra virginica
	Bolboschoenus fluviatilis
	Vitis riparia
	Impatiens capensis
	Hibiscus moscheutos
	Leersia oryzoides
	Polygonum arifolium
	Calystegia sepium
	Sagittaria latifolia
	Cephalanthus occidentalus
	Bidens spp
	Acorus calamus
	Pontederia cordata
	Nuphar lutea
	Rosa multiflora
	Mikania scandens
	Zizania aquatica

	Bidens laevis
	Amaranthus cannabinus
	Grass spp
	Polygonum punctatum
	Schoenoplectus tabernaemontani
	Pilea pumila
Dyke Marsh West	*Nuphar lutea*
	Pontederia cordata
	Typha angustifolia
	Acorus calamus
	Peltandra virginica
	Impatiens capensis
	Leersia oryzoides
	Apios americana
	Vigna luteola
	Bolboschoenus fluviatilis
	Cephalanthus occidentalus
	Zizania aquatica
	Hibiscus moscheutos
	Acer rubrum
	Rosa multiflora
	Sambucus nigra ssp canadensis
	Rosa palustris
	Polygonum arifolium
	Clematis crispa
Dyke Marsh South	*Typha angustifolia*
	Impatiens capensis
	Leersia oryzoides
	Peltandra virginica
	Mikania scandens
	Apios americana
	Bolboschoenus fluviatilis
	Acorus calamus
	Cephalanthus occidentalus
	Nuphar lutea
	Pontederia cordata
	Zizania aquatica
	Solanum dulcamara
	Hibiscus moscheutos
	Bidens laevis
	Sagittaria latifolia
	Polygonum arifolium
Featherstone NWR South	*Phragmites australis (invasive)*

	Nuphar lutea
	Peltandra virginica
	Pontederia cordata
	Zizania aquatica
	Eleocharis quadrangulata
	Schoenoplectus americanus
	Apios americana
	Typha angustifolia
	Rosa palustris
	Impatiens capensis
	Boehmeria cylindrica
	Onoclea sensibilis
	Thelypteris palustris
	Polygonum lapathafolium
	Amaranthus cannabinus
	Sagittaria latifolia
	Mikania scandens
	Leersia oryzoides
	Sparganium americanum
	Parthenocissus quinquefolia
	Arthraxon hispidus
	Lycopus virginicus
	Hibiscus moscheutos
	Polygonum arifolium
	Typha latifolia
	Polygonum spp
	Asclepias incarnata
	Decodon verticillatus
	Bolboschoenus fluviatilis
	Sambucus nigra ssp canadensis
	Pilea pumila
	Calystegia sepium
	Echinocloa muricata
	Alternanthera philoxeroides
Featherstone NWR Central	*Justicia americana*
	Boehmeria cylindrica
	Polygonum arifolium
	Typha angustifolia
	Apios americana
	Leersia oryzoides
	Bolboschoenus fluviatilis
	Calystegia sepium
	Solanum dulcamara

Rumex verticillatus
Decodon verticillatus
Urtica dioica
Peltandra virginica
Polygonum sagittatum
Commelina virginica
Mikania scandens
Carex spicata
Sambucus nigra ssp canadensis
Amorpha fruticosa
Vigna luteola
Sparganium americanum
Polygonum spp
Pilea pumila
Bidens spp
Hibiscus moscheutos
Sagittaria latifolia
Onoclea sensibilis
Pontederia cordata
Schoenoplectus tabernaemontani
Juncus effusus
Stachys tenuifolia
Carex comosa
Impatiens capensis
Eleocharis spp
Carex spp
Toxicodendron radicans
Thelypteris palustris
Lathyrus palustris
Schoenoplectus americanus
Schoenoplectus pungens
Typha latifolia
Murdannia keisak
Panicum dichotomiflorum
Polygonum hydropiperoides
Saccharum giganteum
Pluchea camphorata
Asclepias incarnata
Scirpus cyperinus
Helianthus tuberosus
Carex lurida
Schoenoplectus acutus
Kosteletzkya virginica

	Aster vimineus
	Lycopus virginicus
	Sium suave
	Verbena hastata
	Phragmites australis (invasive)
Featherstone NWR North	*Gratiola virginiana*
	Galium tinctorium
	Phragmites australis (invasive)
	Apios americana
	Typha angustifolia
	Onoclea sensibilis
	Polygonum arifolium
	Rumex verticillatus
	Impatiens capensis
	Polygonum spp
	Amorpha fruticosa
	Cephalanthus occidentalus
	Vigna luteola
	Peltandra virginica
	Thelypteris palustris
	Stachys hyssopifolia
	Schoenoplectus americanus
	Hibiscus moscheutos
	Bidens spp
	Sagittaria latifolia
	Pontederia cordata
	Nuphar lutea
	Eleocharis spp
	Leersia oryzoides
	Schoenoplectus tabernaemontani
	Decodon verticillatus
	Polygonum sagittatum
	Boehmeria cylindrica
	Grass spp
	Rosa multiflora
	Galium spp
	Carex comosa
	Mikania scandens
	Vitis riparia
	Pluchea camphorata
	Sparganium americanum
	Bolboschoenus fluviatilis
	Carex lurida

	Parthenocissus quinquefolia
	Aster vimineus
	Pilea pumila
	Kosteletzkya virginica
	Typha latifolia
	Carex albolutescens
	Salix nigra
	Rosa palustris
	Acer rubrum
	Sambucus nigra ssp canadensis
	Toxicodendron radicans
Featherstone NWR Creek	*Phragmites australis (invasive)*
	Typha angustifolia
	Bolboschoenus fluviatilis
	Cephalanthus occidentalus
	Thelypteris palustris
	Rumex verticillatus
	Cornus amomum
	Hibiscus moscheutos
	Peltandra virginica
	Decodon verticillatus
	Mikania scandens
	Impatiens capensis
	Sagittaria latifolia
	Boehmeria cylindrica
	Galium tinctorium
	Carex lacustris
	Vitis riparia
	Leersia oryzoides
	Nuphar lutea
	Pontederia cordata
	Schoenoplectus americanus
	Carex comosa
	Schoenoplectus acutus
	Stachys hyssopifolia
	Polygonum arifolium
	Bidens spp
	Schoenoplectus pungens
	Typha latifolia
	Sparganium americanum
	Bidens laevis
	Polygonum hydropiperoides
	Sambucus nigra ssp canadensis

	Onoclea sensibilis
	Murdannia keisak
	Apios americana
	Amaranthus cannabinus
Fort Belvoir Jack Abbott	*Sagittaria latifolia*
	Nuphar lutea
	Leersia oryzoides
	Sparganium americanum
	Heteranthera reniformis
	Bidens spp
	Eleocharis obtusa
	Bidens laevis
	Polygonum spp
	Ludwigia alternifolia
	Hibiscus moscheutos
	Juncus effusus
	Arthraxon hispidus
	Scirpus cyperinus
	Salix nigra
	Pilea pumila
	Triadenum virgincum
	Cephalanthus occidentalus
	Mikania scandens
	Decodon verticillatus
	Vigna luteola
	Galium trifidum
	Grass spp
	Boehmeria cylindrica
	Peltandra virginica
	Clematis crispa
	Polygonum hydropiperoides
	Impatiens capensis
	Carex comosa
	Polygonum setaceum
	Cirsium spp
	Murdannia keisak
	Polygonum sagittatum
	Polygonum lapathafolium
	Apios americana
	Calystegia sepium
	Galium tinctorium
	Cyperus strigosus
	Cyperus erythrorhizos

	Hypericum mutilum
	Eleocharis tuberculosa
	Eleocharis parvula
	Ammannia latifolia
	Ludwigia palustris
	Lindernia dubia
	Typha latifolia
Fort Belvoir Loop Trail	*Peltandra virginica*
	Juncus effusus
	Hibiscus moscheutos
	Scirpus cyperinus
	Salix nigra
	Leersia oryzoides
	Pilea pumila
	Murdannia keisak
	Carex lurida
	Carex comosa
	Cephalanthus occidentalus
	Saururus cernuus
	Panicum hemitomon
	Boehmeria cylindrica
	Typha latifolia
	Bidens laevis
Fort Belvoir Wetlands Trail	*Pontederia cordata*
	Peltandra virginica
	Sagittaria latifolia
	Hibiscus moscheutos
	Juncus effusus
	Typha angustifolia
	Cephalanthus occidentalus
	Scirpus cyperinus
	Hydracotyle ranunculoides
	Ludwigia palustris
	Leersia oryzoides
	Apios americana
	Cyperus erythrorhizos
	Alisma subcordatum
	Bidens spp
	Dulichium arundinaceum
	Schoenoplectus acutus
	Polygonum spp
	Ranunculus ambigens
	Sparganium americanum

	Rhynchospora macrostachya
	Carex comosa
	Saururus cernuus
	Pilea pumila
	Eleocharis obtusa
	Orontium aquaticum
	Bolboschoenus fluviatilis
	Rosa palustris
	Murdannia keisak
	Typha latifolia
	Scutellaria lateriflora
	Acer rubrum
	Boehmeria cylindrica
	Galium tinctorium
	Vigna luteola
	Echinocloa muricata
	Conyza canadensis
	Osmunda regalis
Fort Belvoir Accotink Marsh	*Zizania aquatica*
	Pontederia cordata
	Peltandra virginica
	Bidens laevis
	Bolboschoenus fluviatilis
	Nuphar lutea
	Sagittaria latifolia
	Typha latifolia
	Phragmites australis (native)
	Leersia oryzoides
	Impatiens capensis
	Schoenoplectus tabernaemontani
	Polygonum arifolium
	Apios americana
	Sium suave
	Echinocloa muricata
	Polygonum sagittatum
	Scutellaria lateriflora
	Hibiscus moscheutos
	Carex lurida
	Sparganium americanum
	Galium tinctorium
	Helenium autumnale
Fort Belvoir Accotink Bay	*Vitis riparia*
	Scirpus cyperinus

	Polygonum arifolium
	Impatiens capensis
	Leersia oryzoides
	Peltandra virginica
	Sparganium americanum
	Polygonum sagittatum
	Boehmeria cylindrica
	Orontium aquaticum
	Carex comosa
	Galium tinctorium
	Mikania scandens
	Apios americana
	Vigna luteola
	Symphyotrichum lanceolatum
	Clematis crispa
	Typha latifolia
	Schoenoplectus tabernaemontani
	Phragmites australis (invasive)
	Sagittaria latifolia
	Carex vulpinodea
	Carex lurida
	Juncus effusus
	Bidens spp
	Murdannia keisak
	Pontederia cordata
	Zizania aquatica
	Nuphar lutea
	Polygonum spp
	Carex conjuncta
	Bidens laevis
	Rumex verticillatus
	Lycopus virginicus
	Schoenoplectus acutus
	Typha angustifolia
	Cyperus strigosus
	Fraxinus pensylvanica
	Hibiscus moscheutos
	Amaranthus cannabinus
Huntley Meadows Marsh Inlet	*Polygonum lapathifolium*
	Juncus effusus
	Hibiscus moscheutos
	Leersia oryzoides
	Cephalanthus occidentalus

	Galium tinctorium
	Typha latifolia
	Hydracotyle ranunculoides
	Scirpus cyperinus
	Bidens spp
	Polygonum hydropiper
	Sagittaria latifolia
	Triadenum virgincum
	Acer rubrum
	Rosa multiflora
	Saururus cernuus
	Galium trifidum
	Ludwigia palustris
	Sambucus nigra ssp canadensis
	Polygonum spp
	Eleocharis spp
	Ludwigia peploides
	Sparganium americanum
	Liquidambar styraciflua
	Echinocloa muricata
	Juncus canadensis
Huntley Meadows Central Marsh	*Hibiscus moscheutos*
	Juncus effusus
	Hydracotyle ranunculoides
	Ludwigia palustris
	Leersia oryzoides
	Acer rubrum
	Rosa palustris
	Lycopus virginicus
	Triadenum virgincum
	Asclepias incarnata
	Polygonum spp
	Bidens spp
	Typha latifolia
	Eleocharis tuberculosa
	Grass spp
	Eleocharis parvula
	Polygonum lapathifolium
	Saururus cernuus
	Eleocharis spp
	Murdannia keisak
	Decodon verticillatus
Huntley Meadows Creek East	*Sagittaria latifolia*

	Hydracotyle ranunculoides
	Typha latifolia
	Bidens spp
	Polygonum hydropiper
	Triadenum virgincum
	Saururus cernuus
	Boehmeria cylindrica
	Rubus allegheniensis
	Ludwigia palustris
	Alisma subcordatum
	Arthraxon hispidus
	Grass spp
	Scirpus cyperinus
	Leersia oryzoides
	Asclepias incarnata
	Carex lurida
	Hibiscus moscheutos
	Nuphar lutea
	Sparganium americanum
	Polygonum spp
	Murdannia keisak
	Polygonum lapathifolium
	Lindernia dubia
	Cephalanthus occidentalus
	Juncus effusus
	Echinocloa muricata
	Acer rubrum
Huntley Meadows Creek West	*Nuphar lutea*
	Typha latifolia
	Saururus cernuus
	Impatiens capensis
	Polygonum sagittatum
	Bidens spp
	Leersia oryzoides
	Sagittaria latifolia
	Cyperus erythrorhizos
	Heteranthera reniformis
	Eleocharis obtusa
	Alisma subcordatum
	Grass spp
	Ludwigia palustris
	Polygonum hydropiper
	Lindernia dubia

	Typha angustifolia
	Boehmeria cylindrica
	Hibiscus moscheutos
	Triadenum virgincum
	Scirpus cyperinus
	Acer rubrum
	Cephalanthus occidentalus
	Carex lurida
	Galium tinctorium
	Echinocloa muricata
	Polygonum spp
	Polygonum lapathifolium
	Hydracotyle ranunculoides
	Bidens laevis
	Carex vulpinodea
	Juncus effusus
	Scutellaria lateriflora
	Peltandra virginica
	Mimulus alatus
	Mikania scandens
	Murdannia keisak
Kenilworth Gardens River Trail	*Typha angustifolia*
	Peltandra virginica
	Schoenoplectus tabernaemontani
	Saururus cernuus
	Leersia oryzoides
	Impatiens capensis
	Pontederia cordata
	Amaranthus cannabinus
	Lythrum salicaria
Kenilworth Gardens Boardwalk	*Pontederia cordata*
	Sagittaria latifolia
	Typha latifolia
	Peltandra virginica
	Zizania aquatica
	Leersia oryzoides
	Carex spp
	Bolboschoenus fluviatilis
	Lythrum salicaria
	Rumex verticillatus
	Typha angustifolia
	Mikania scandens
	Saururus cernuus

	Schoenoplectus tabernaemontani
	Nuphar lutea
	Polygonum arifolium
	Impatiens capensis
	Amaranthus cannabinus
	Bidens laevis
Kingman Marsh South	*Peltandra virginica*
	Nuphar lutea
	Brasenia schreberi
	Pontederia cordata
	Zizania aquatica
	Typha angustifolia
	Schoenoplectus tabernaemontani
	Polygonum punctatum
	Lythrum salicaria
	Bidens laevis
	Iris pseudacorus
	Typha latifolia
Kingman Marsh North	*Peltandra virginica*
	Pontederia cordata
	Nuphar lutea
	Typha latifolia
	Sagittaria latifolia
	Lythrum salicaria
	Polygonum hydropiper
	Iris spp
	Typha angustifolia
	Phragmites australis (invasive)
	Polygonum punctatum
	Iris pseudacorus
	Bidens laevis
	Salix nigra
	Amaranthus cannabinus
	Schoenoplectus tabernaemontani
	Impatiens capensis
Mason Neck NWR Great Marsh West	*Peltandra virginica*
	Sparganium americanum
	Bidens laevis
	Impatiens capensis
	Leersia oryzoides
	Polygonum arifolium
	Typha latifolia
	Cephalanthus occidentalus

	Polygonum punctatum
	Amaranthus cannabinus
	Polygonum virginianum
	Mikania scandens
	Sagittaria latifolia
	Bolboschoenus fluviatilis
	Hibiscus moscheutos
	Eleocharis obtusa
	Boehmeria cylindrica
	Decodon verticillatus
	Typha angustifolia
	Acorus calamus
	Salix nigra
	Aster vimineus
	Solanum dulcineum
	Onoclea sensibilis
	Eleocharis tuburculosa
	Carex lurida
	Polygonum spp
	Carex comosa
	Mentha arvensis
	Zizania aquatica
	Rumex verticillatus
	Nuphar lutea
Mason Neck NWR Great Marsh Central	*Zizania aquatica*
	Peltandra virginica
	Sparganium americanum
	Pontederia cordata
	Typha angustifolia
	Bidens laevis
	Amaranthus cannabinus
	Polygonum spp
	Phragmites australis (native)
	Mikania scandens
	Impatiens capensis
	Bolboschoenus fluviatilis
	Rumex verticillatus
	Leersia oryzoides
	Clematis crispa
	Hibiscus moscheutos
	Apios americana
	Sagittaria latifolia
	Sambucus nigra ssp canadensis

	Polygonum arifolium
	Boehmeria cylindrica
	Salix nigra
	Schoenoplectus acutus
	Nuphar lutea
Mason Neck NWR Great Marsh North	*Nuphar lutea*
	Peltandra virginica
	Zizania aquatica
	Schoenoplectus tabernaemontani
	Bolboschoenus fluviatilis
	Hibiscus moscheutos
	Rumex verticillatus
	Impatiens capensis
	Leersia oryzoides
	Typha angustifolia
	Phragmites australis (native)
	Clematis crispa
	Pontederia cordata
	Sagittaria latifolia
	Amaranthus cannabinus
	Bidens laevis
	Sium suave
	Schoenoplectus acutus
	Mikania scandens
	Apios americana
	Boehmeria cylindrica
Mason Neck NWR Great Marsh East	*Nuphar lutea*
	Pontederia cordata
	Peltandra virginica
	Typha angustifolia
	Bolboschoenus fluviatilis
	Schoenoplectus tabernaemontani
	Decodon verticillatus
	Iris spp
	Sparganium americanum
	Polygonum spp
	Phragmites australis (native)
	Boehmeria cylindrica
	Onoclea sensibilis
	Impatiens capensis
	Leersia oryzoides
	Clematis crispa
	Rumex verticillatus

	Rosa palustris
	Salix nigra
	Hibiscus moscheutos
	Sambucus nigra ssp canadensis
	Lysimachia vulgaris
	Cicuta maculata
	Carex spp
	Apios americana
	Bidens laevis
	Amaranthus cannabinus
	Mikania scandens
	Schoenoplectus acutus
	Zizania aquatica
	Acorus calamus
	Lamiaceae spp
	Urtica dioica
	Polygonum arifolium
Mason Neck State Park East	*Peltandra virginica*
	Sagittaria latifolia
	Zizania aquatica
	Bidens spp
	Typha angustifolia
	Bolboschoenus fluviatilis
	Mikania scandens
	Impatiens capensis
	Leersia oryzoides
	Polygonum arifolium
	Polygonum spp
	Vernonia noveboracensis
	Cicuta maculata
	Eleocharis spp
	Asclepias incarnata
	Dulichium arundinaceum
	Murdannia keisak
	Rosa multiflora
	Polygonum sagittatum
	Pontederia cordata
	Bidens laevis
	Polygonum hydropiper
	Hibiscus moscheutos
	Cephalanthus occidentalus
	Sium suave
	Schoenoplectus tabernaemontani

	Parthenocissus quinquefolia
	Eleocharis obtusa
	Rosa palustris
	Orontium aquaticum
	Ludwigia palustris
	Typha latifolia
Mason Neck State Park Central	*Nuphar lutea*
	Peltandra virginica
	Zizania aquatica
	Bidens spp
	Typha angustifolia
	Leersia oryzoides
	Mikania scandens
	Decodon verticillatus
	Sagittaria latifolia
	Polygonum hydropiperoides
	Boehmeria cylindrica
	Pilea pumila
	Polygonum arifolium
	Clematis crispa
	Cicuta maculata
	Symphyotrichum prenanthoides
	Saururus cernuus
	Murdannia keisak
	Galium spp
	Pontederia cordata
	Polygonum spp
	Carex lurida
	Bidens laevis
	Polygonum hydropiper
	Sium suave
	Acorus calamus
	Ludwigia palustris
	Apios americana
	Aster vimineus
Mason Neck State Park West	*Nuphar lutea*
	Peltandra virginica
	Zizania aquatica
	Pontederia cordata
	Typha angustifolia
	Acorus calamus
	Schoenoplectus tabernaemontani
	Bidens spp

	Bolboschoenus fluviatilis
	Carex comosa
	Leersia oryzoides
	Impatiens capensis
	Sparganium americanum
	Eleocharis quadrangulata
	Fraxinus pensylvanica
	Mikania scandens
	Polygonum arifolium
	Carex vulpinodea
	Carex lurida
	Boehmeria cylindrica
	Sagittaria latifolia
	Schoenoplectus pungens
	Bidens laevis
	Schoenoplectus acutus
	Polygonum hydropiper
	Pilea pumila
	Saururus cernuus
	Typha latifolia
Mattawoman Creek West	*Nelumbo lutea*
	Nuphar lutea
	Peltandra virginica
	Zizania aquatica
	Pontederia cordata
	Bidens laevis
	Typha angustifolia
	Sagittaria latifolia
	Cyperus spp
	Leersia oryzoides
	Decodon verticillatus
	Hibiscus moscheutos
	Schoenoplectus tabernaemontani
	Sium suave
	Bolboschoenus fluviatilis
	Schoenoplectus acutus
	Ludwigia palustris
	Amaranthus cannabinus
Mattawoman Creek Island	*Peltandra virginica*
	Zizania aquatica
	Nuphar lutea
	Pontederia cordata
	Sagittaria latifolia

	Bidens laevis
	Typha angustifolia
	Hibiscus moscheutos
	Leersia oryzoides
	Amaranthus cannabinus
	Impatiens capensis
	Decodon verticillatus
	Polygonum spp
	Ludwigia palustris
	Heteranthera reniformis
	Mikania scandens
	Asclepias incarnata
	Lobelia cardinalis
	Murdannia keisak
	Sium suave
Mattawoman Creek Central	*Peltandra virginica*
	Zizania aquatica
	Nuphar lutea
	Sagittaria latifolia
	Schoenoplectus pungens
	Bidens laevis
	Typha angustifolia
	Pontederia cordata
	Cyperus spp
	Eleocharis tuberculosa
	Decodon verticillatus
	Schoenoplectus tabernaemontani
	Hibiscus moscheutos
	Leersia oryzoides
	Mikania scandens
	Bolboschoenus fluviatilis
	Eleocharis quadrangulata
	Schoenoplectus acutus
	Polygonum sagittatum
	Amaranthus cannabinus
	Sparganium americanum
	Impatiens capensis
	Sium suave
	Carex lurida
	Galium tinctorum
	Polygonum spp
	Polygonum arifolium
	Eleocharis obtusa

	Carex stricta
	Scirpus cyperinus
	Polygonum setaceum
	Apios americana
	Boehmeria cylindrica
	Vigna luteola
	Rosa palustris
	Asclepias incarnata
	Acer rubrum
Mattawoman Creek North	*Peltandra virginica*
	Nuphar lutea
	Zizania aquatica
	Hibiscus moscheutos
	Leersia oryzoides
	Bidens laevis
	Asclepias incarnata
	Mikania scandens
	Impatiens capensis
	Rosa palustris
	Typha angustifolia
	Pontederia cordata
	Galium obtusum
	Boehmeria cylindrica
	Decodon verticillatus
	Schoenoplectus pungens
	Thelypteris palustris
	Cephalanthus occidentalus
	Ludwigia palustris
	Murdannia keisak
	Amaranthus cannabinus
	Polygonum arifolium
	Lamiaceae spp
	Lobelia cardinalis
	Galium tinctorum
	Pilea pumila
	Eleocharis obtusa
	Carex comosa
	Sagittaria latifolia
	Apios americana
	Elymus virginicus
	Carex stricta
	Cinna arundinacea
	Cornus amomum

	Polygonum setaceum
Mattawoman Creek East	*Nuphar lutea*
	Peltandra virginica
	Impatiens capensis
	Mikania scandens
	Pontederia cordata
	Galium obtusum
	Typha angustifolia
	Polygonum spp
	Sium suave
	Cephalanthus occidentalus
	Asclepias incarnata
	Bidens laevis
	Leersia oryzoides
	Zizania aquatica
	Sagittaria latifolia
	Hibiscus moscheutos
	Ludwigia palustris
	Lobelia cardinalis
	Cuscuta gronovii
	Polygonum setaceum
	Amaranthus cannabinus
	Sparganium americanum
	Polygonum arifolium
	Bidens spp
	Juncus canadensis
	Scirpus cyperinus
	Helenium autumnale
	Carex comosa
	Kosteletzkya virginica
	Juncus effusus
	Symphyotrichum puniceum
	Carex lurida
	Polygonum sagittatum
Neabsco Creek West	*Typha latifolia*
	Nuphar lutea
	Juncus effusus
	Polygonum arifolium
	Polygonum hydropiper
	Bidens spp
	Polygonum pennsylvanicum
	Heteranthera reniformis
	Helenium autumnale

	Eleocharis tuberculosa
	Echinochloa muricata
	Leersia oryzoides
	Polygonum sagittatum
	Carex spp
	Carex lurida
	Ammannia latifolia
	Grass spp
	Murdannia keisak
	Alisma subcordatum
	Impatiens capensis
	Hibiscus moscheutos
	Peltandra virginica
	Mikania scandens
	Sparganium americanum
	Apios americana
	Calystegia sepium
	Acer rubrum
	Sambucus nigra ssp canadensis
	Cephalanthus occidentalus
	Polygonum spp
	Polygonum lapathifolium
	Cyperus strigosus
	Rumex verticillatus
	Schoenoplectus tabernaemontani
	Cuscuta gronovii
	Polygonum perfoliatum
	Pilea pumila
	Carex comosa
	Onoclea sensibilis
	Scirpus cyperinus
	Eupatorium serotinum
	Verbena hastata
Neabsco Creek North	*Sparganium americanum*
	Nuphar lutea
	Pontederia cordata
	Murdannia keisak
	Grass spp
	Polygonum sagittatum
	Polygonum spp
	Vigna luteola
	Typha latifolia
	Peltandra virginica

	Onoclea sensibilis
	Impatiens capensis
	Polygonum arifolium
	Apios americana
	Boehmeria cylindrica
	Mikania scandens
	Carex spp
	Hibiscus moscheutos
	Leersia oryzoides
	Vernonia noveboracensis
	Galium spp
	Eleocharis obtusa
	Polygonum lapathifolium
	Schoenoplectus acutus
	Echinochloa muricata
	Polygonum hydropiperoides
	Sagittaria latifolia
	Bolboschoenus fluviatilis
	Bidens spp
	Solidago spp
	Asclepias incarnata
	Carex lurida
	Pilea pumila
Neabsco Creek Central	*Nuphar lutea*
	Zizania aquatica
	Pontederia cordata
	Peltandra virginica
	Sagittaria latifolia
	Typha angustifolia
	Boehmeria cylindrica
	Leersia oryzoides
	Amaranthus cannabinus
	Polygonum arifolium
	Impatiens capensis
	Mikania scandens
	Rorippa palustris
	Typha latifolia
	Pilea pumila
	Polygonum sagittatum
Neabsco Creek South	*Zizania aquatica*
	Nuphar lutea
	Pontederia cordata
	Typha angustifolia

	Peltandra virginica
	Polygonum arifolium
	Amaranthus cannabinus
	Sagittaria latifolia
	Polygonum spp
	Impatiens capensis
	Typha latifolia
	Leersia oryzoides
Occoquan Bay Wildlife Drive	*Peltandra virginica*
	Leersia oryzoides
	Sparganium americanum
	Hibiscus moscheutos
	Boehmeria cylindrica
	Murdannia keisak
	Cephalanthus occidentalus
	Juncus effusus
	Scirpus cyperinus
	Pontederia cordata
	Eleocharis parvula
	Ludwigia palustris
	Nuphar lutea
	Cladium mariscoides
	Decodon verticillatus
	Typha latifolia
	Sagittaria latifolia
	Sambucus nigra ssp canadensis
	Pilea pumila
	Bidens spp
	Eleocharis palustris
	Heteranthera reniformis
	Galium tinctorium
	Carex vulpinoidea
	Carex comosa
	Eleocharis tuberculosa
	Alisma subcordatum
	Apios americana
	Carex scoparia
	Polygonum arifolium
	Juncus canadensis
	Carex lurida
	Onoclea sensibilis
	Schoenoplectus americanus
	Lindernia dubia

Occoquan Bay Deephole Point North	*Toxicodendron radicans*
	Boehmeria cylindrica
	Sparganium americanum
	Clematis crispa
	Justicia americana
	Juncus effusus
	Peltandra virginica
	Apios americana
	Scirpus cyperinus
	Murdannia keisak
	Pontederia cordata
	Hibiscus moscheutos
	Carex lurida
	Rubus allegheniensis
	Vitis riparia
	Rosa palustris
	Rumex verticillatus
	Sambucus nigra ssp canadensis
	Cephalanthus occidentalus
	Asclepias incarnata
	Cyperus erythrorhizos
	Prunus americana
	Saururus cernuus
	Nuphar lutea
	Sagittaria latifolia
	Eleocharis palustris
	Leersia oryzoides
	Typha latifolia
	Decodon verticillatus
	Polygonum spp
	Galium trifidum
	Carex comosa
	Mikania scandens
	Bolboschoenus fluviatilis
	Schoenoplectus tabernaemontani
	Salix nigra
	Ludwigia palustris
	Aster vimineus
	Pilea pumila
	Carex vulpinoidea
	Ulmus americana
	Lindernia dubia
	Polygonum punctatum

	Eleocharis tuberculosa
	Juncus canadensis
Occoquan Bay Deephole Point South	*Hibiscus moscheutos*
	Cephalanthus occidentalus
	Bolboschoenus fluviatilis
	Peltandra virginica
	Saururus cernuus
	Leersia oryzoides
	Sagittaria latifolia
	Acorus calamus
	Clematis crispa
	Rumex verticillatus
	Boehmeria cylindrica
	Nuphar lutea
	Pontederia cordata
	Typha angustifolia
	Decodon verticillatus
	Sparganium americanum
	Impatiens capensis
	Polygonum arifolium
	Amaranthus cannabinus
	Murdannia keisak
	Pilea pumila
	Schoenoplectus tabernaemontani
	Bidens spp
	Aster vimineus
	Calystegia sepium
	Asclepias incarnata
Occoquan Bay Marumsco Creek	*Impatiens capensis*
	Typha latifolia
	Peltandra virginica
	Leersia oryzoides
	Hibiscus moscheutos
	Mikania scandens
	Sparganium americanum
	Acorus calamus
	Toxicodendron radicans
	Polygonum arifolium
	Murdannia keisak
	Aster spp
	Pilea pumila
	Nuphar lutea
	Zizania aquatica

	Pontederia cordata
	Apios americana
	Rumex verticillatus
	Scirpus cyperinus
	Cephalanthus occidentalus
	Sagittaria latifolia
	Typha angustifolia
	Juncus effusus
	Schoenoplectus americanus
	Lysimachia terrestris
	Thelypteris palustris
	Saururus cernuus
	Prunus americana
	Vitis riparia
	Boehmeria cylindrica
	Vigna luteola
	Sium suave
	Cuscuta gronovii
	Amaranthus cannabinus
	Carex lurida
	Carex comosa
	Aster vimineus
	Itea virginica
	Sambucus nigra ssp canadensis
	Polygonum sagittatum
	Fraxinus pennsylvanica
	Decodon verticillatus
	Calystegia sepium
	Polygonum spp
	Polygonum setaceum
Patuxent Refuge Rogue Harbor	*Nuphar lutea*
	Heteranthera reniformis
	Murdannia keisak
	Eleocharis obtusa
	Sparganium americanum
	Ludwigia palustris
	Alisma subcordatum
	Polygonum lapathifolium
	Ludwigia peploides
	Echinochloa muricata
	Scirpus cyperinus
	Saururus cernuus
	Mikania scandens

	Leersia oryzoides
	Typha latifolia
	Boehmeria cylindrica
	Onoclea sensibilis
	Galium obtusum
	Cuscuta gronovii
	Pilea pumila
	Polygonum sagittatum
	Hibiscus moscheutos
	Juncus effusus
	Potamogeton pulcher
	Schoenoplectus tabernaemontani
	Sagittaria latifolia
	Ludwigia sphaerocarpa
	Galium tinctorium
	Polygonum arifolium
	Carex comosa
	Polygonum pensylvanicum
	Polygonum hydropiperoides
	Bidens spp
	Arthraxon hispidus
	Verbena hastata
Patuxent Refuge Uhler Marsh	*Juncus effusus*
	Juncus marginatus
	Eleocharis quadrangulata
	Eupatorium spp
	Leersia oryzoides
	Ludwigia alternifolia
	Triadenum virgincum
	Conyza canadensis
	Sagittaria latifolia
	Juncus spp
	Galium trifidum
	Hypericum mutilum
	Grass spp
	Carex spp
	Rosa multiflora
	Microstegium vimineum
	Asclepias incarnata
	Juncus coriaceus
	Bidens spp
	Rhynchospora macrostachya
	Polygonum punctatum

	Juncus canadensis
	Eleocharis spp
	Xyris difformis
	Acer rubrum
	Ludwigia palustris
	Nuphar lutea
	Potamogeton pulcher
	Sparganium americanum
	Typha angustifolia
	Iris pseudacorus
	Decodon verticillatus
	Brasenia schreberi
	Rhexia virginica
	Polygonum hydropiperoides
	Typha latifolia
	Scirpus cyperinus
	Cephalanthus occidentalus
	Salix nigra
	Carex comosa
	Cyperus refractus
	Polygonum spp
	Cypeurs pseudovegetus
	Eleocharis tuberculosa
	Liquidambar styraciflua
	Dulichium arundinaceum
	Echinochloa muricata
Patuxent Refuge Knowles North	*Typha latifolia*
	Juncus effusus
	Decodon verticillatus
	Cephalanthus occidentalus
	Polygonum punctatum
	Ludwigia palustris
	Nuphar lutea
	Galium trifidum
	Salix nigra
	Rhynchospora macrostachya
	Eleocharis quadrangulata
	Polygonum hydropiperoides
	Scirpus cyperinus
	Acer rubrum
	Typha angustifolia
	Triadenum virgincum
Patuxent Refuge Knowles South	*Nymphaea odorata*

	Nuphar lutea
	Typha latifolia
	Eleocharis quadrangulata
	Decodon verticillatus
	Cephalanthus occidentalus
	Polygonum hydropiperoides
	Galium obtusum
	Acer rubrum
	Ludwigia palustris
	Hibiscus moscheutos
	Juncus effusus
	Triadenum virgincum
	Carex crinita
	Lysimachia terrestris
	Sparganium americanum
	Dulichium arundinaceum
	Rhynchospora macrostachya
	Salix nigra
Patuxent Refuge Lake Redington	*Nuphar lutea*
	Typha latifolia
	Juncus effusus
	Leersia oryzoides
	Carex lurida
	Galium trifidum
	Dulichium arundinaceum
	Polygonum punctatum
	Boehmeria cylindrica
	Lycopus virginicus
	Triadenum virgincum
	Sagittaria latifolia
	Ludwigia palustris
	Eleocharis quadrangulata
	Galium obtusum
	Acer rubrum
	Cyperus spp
	Grass spp
	Bidens spp
	Nymphaea odorata
	Brasenia schreberi
	Scirpus cyperinus
	Carex comosa
	Cyperus pseudovegetus
	Galium tinctorium

	Hypericum mutilum
	Polygonum hydropiperoides
	Eleocharis tuberculosa
	Cephalanthus occidentalus
	Eleocharis spp
Powells Creek East	*Nuphar lutea*
	Pontederia cordata
	Zizania aquatica
	Peltandra virginica
	Impatiens capensis
	Typha angustifolia
	Schoenoplectus tabernaemontani
	Polygonum arifolium
	Mikania scandens
	Leersia oryzoides
	Bolboschoenus fluviatilis
	Lysimachia terrestris
	Asclepias incarnata
	Carex vulpinodea
	Lathyrus palustris
	Solidago spp
	Hibiscus moscheutos
	Schoenoplectus americanus
	Apios americana
	Galium spp
	Eleocharis quadrangulata
	Eleocharis spp.
	Dulichium arundinaceum
	Acorus calamus
	Boehmeria cylindrica
	Acer rubrum
	Rubus idaeus ssp strigosus
	Rosa palustris
	Vigna luteola
	Echinocloa muricata
	Sagittaria latifolia
	Bidens laevis
	Scirpus cyperinus
	Eleocharis tuburculosa
	Polygonum spp
	Carex comosa
	Murdannia keisak
	Carex stricta

	Betula nigra
	Cephalanthus occidentalus
	Polygonum sagittatum
Powells Creek Central	*Phragmites australis (native)*
	Peltandra virginica
	Typha angustifolia
	Impatiens capensis
	Pontederia cordata
	Cephalanthus occidentalus
	Nuphar lutea
	Hibiscus moscheutos
	Zizania aquatica
	Bidens laevis
	Sagittaria latifolia
	Schoenoplectus tabernaemontani
	Calystegia sepium
	Bolboschoenus fluviatilis
	Schoenoplectus americanus
	Leersia oryzoides
	Typha latifolia
	Amaranthus cannabinus
	Vigna luteola
Powells Creek South	*Nuphar lutea*
	Peltandra virginica
	Pontederia cordata
	Typha angustifolia
	Leersia oryzoides
	Hibiscus moscheutos
	Bolboschoenus fluviatilis
	Schoenoplectus tabernaemontani
	Bidens laevis
	Eleocharis spp
	Sparganium americanum
	Cephalanthus occidentalus
	Justicia americana
	Eleocharis quadrangulata
	Sagittaria latifolia
	Zizania aquatica
	Mikania scandens
	Murdannia keisak
	Saururus cernuus
	Polygonum spp
	Commelina virginica

	Lobelia cardinalis
	Salix nigra
	Echinocloa muricata
	Rumex verticillatus
	Carex stricta
	Aster vimineus
	Symphyotrichum puniceum
	Polygonum arifolium
Powells Creek West	*Nuphar lutea*
	Saururus cernuus
	Bolboschoenus fluviatilis
	Peltandra virginica
	Leersia oryzoides
	Mikania scandens
	Boehmeria cylindrica
	Grass spp
	Hibiscus moscheutos
	Galium spp
	Sagittaria latifolia
	Schoenoplectus tabernaemontani
	Typha angustifolia
	Carex spp
	Polygonum arifolium
	Apios americana
	Cephalanthus occidentalus
	Ulmus americana
	Zizania aquatica
	Acorus calamus
	Bidens spp
	Eleocharis parvula
	Polygonum punctatum
	Impatiens capensis
	Iris spp
	Eleocharis quadrangulata
	Pontederia cordata
	Bidens laevis
	Polygonum setaceum
	Sparganium americanum
	Amaranthus cannabinus
	Murdannia keisak
	Sium suave
	Polygonum sagittatum
	Scirpus cyperinus

	Iris virginica
	Rotala ramosior
	Eleocharis spp
	Aster spp
	Eryngium aquaticum
	Pilea pumila
	Vigna luteola
River Terrace Fringe	*Leersia oryzoides*
	Typha angustifolia
	Peltandra virginica
	Mikania scandens
	Sagittaria latifolia
	Pontederia cordata
	Amaranthus cannabinus
	Impatiens capensis
	Lythrum salicaria
	Schoenoplectus tabernaemontani
	Polygonum hydropiper
	Polygonum lapathifolium
	Phragmites australis (invasive)
	Bidens spp
	Polygonum arifolium
	Juncus effusus
	Typha latifolia
	Zizania aquatica
	Bidens laevis
	Polygonum spp

Appendix C: Invertebrate Families Identified at Bird Survey Points

Site Name	Invertebrate Families Identified
Dyke Marsh North	Hirudinidae
	Oligocheates
	Syrphidae
	Asellidae
	Tipulidae
	Lestidae
	Athericidae
	Stratiomyidae
	Chironomidae
	Elmidae
	Hydrophilidae
Dyke Marsh Lagoon	Stratiomyidae
	Chironomidae
	Ceratopogonidae
	Oligocheates
	Belastomatidae
	Lestidae
	Hyallelidae
	Gyrinidae
Dyke Marsh West	Belastomatidae
	Oligocheates
	Syrphidae
	Chironomidae
	Circulionidae
	Gyrinidae
Dyke Marsh South	Stratiomyidae
	Syrphidae
	Hirudinidae
	Oligocheates
	Gammaridae
	Chironomidae
	Lestidae
	Ceratopogonidae
Featherstone NWR South	Belastomatidae
	Hirudinidae
	Gammaridae
	Chironomidae
	Crangonyctidae
	Lestidae

	Asellidae
Featherstone NWR Central	Oligocheates
	Hirudinidae
	Culcidae
	Chironomidae
	Lestidae
Featherstone NWR North	Belastomatidae
	Chironomidae
	Oligocheates
	Culcidae
	Lestidae
	Elmidae
	Gammaridae
	Tabanidae
Featherstone NWR Creek	Oligocheates
	Chironomidae
	Gammaridae
	Crangonyctidae
Fort Belvoir Jack Abbott	Corixidae
	Gammaridae
	Chironomidae
	Oligocheates
	Tipulidae
	Belastomatidae
	Lestidae
	Libellulidae
	Caenidae
	Asellidae
Fort Belvoir Loop Trail	Chironomidae
	Culcidae
	Dysticidae
	Notonectidae
	Tipulidae
	Elmidae
	Belastomatidae
	Corixidae
	Haliplidae
	Ceratopogonidae
	Oligocheates
	Coenagrionidae
	Aeshnidae
Fort Belvoir Wetlands Trail	Corixidae
	Chironomidae

	Oligocheates
	Culcidae
	Syrphidae
	Ceratopogonidae
	Ptychopteridae
	Tipulidae
	Empididae
	Polycentropodidae
	Athericidae
Fort Belvoir Accotink Marsh	Cambaridae
	Oligocheates
	Hirudinidae
	Chironomidae
	Empididae
	Lestidae
	Tabanidae
Fort Belvoir Accotink Bay	Corixidae
	Libellulidae
	Cambaridae
	Chironomidae
	Syrphidae
	Tipulidae
	Dysticidae
Huntley Meadows Marsh Inlet	Belastomatidae
	Chironomidae
	Corixidae
	Elmidae
	Lestidae
	Gammaridae
	Hydrophilidae
	Palaemonidae
Huntley Meadows Central Marsh	Cambaridae
	Belastomatidae
	Lestidae
	Corixidae
	Baetidae
	Libellulidae
	Chironomidae
	Hydrophilidae
	Haliplidae
	Palaemonidae
	Nepidae
	Oligocheates

	Tabanidae
Huntley Meadows Creek East	Belastomatidae
	Hydrophilidae
	Elmidae
	Culcidae
	Tipulidae
	Circulionidae
	Dysticidae
	Palaemonidae
	Chironomidae
	Libellulidae
	Aeshnidae
	Lestidae
Huntley Meadows Creek West	Chironomidae
	Hydrophilidae
	Culcidae
	Belastomatidae
	Corixidae
	Libellulidae
	Dysticidae
	Coenagrionidae
	Baetidae
Kenilworth Gardens River Trail	Syrphidae
	Chironomidae
	Circulionidae
	Belastomatidae
	Hydrachnida
	Gammaridae
	Hirudinidae
	Corixidae
	Oligocheates
Kenilworth Gardens Boardwalk	Syrphidae
	Chironomidae
	Stratiomyidae
	Oligocheates
	Circulionidae
	Asellidae
	Belastomatidae
	Ceratopogonidae
	Lestidae
Kingman Marsh South	Syrphidae
	Chironomidae
	Hirudinidae

	Oligocheates
	Ceratopogonidae
Kingman Marsh North	Syrphidae
	Tipulidae
	Oligocheates
	Ceratopogonidae
	Chironomidae
	Hirudinidae
	Culcidae
Mason Neck NWR Great Marsh West	Hirudinidae
	Belastomatidae
	Circulionidae
	Tricladida
	Chironomidae
	Syrphidae
	Curculionidae
	Dixidae
	Lestidae
	Elmidae
	Gammaridae
Mason Neck NWR Great Marsh Central	Hirudinidae
	Chironomidae
	Syrphidae
	Asellidae
	Elmidae
	Oligocheates
	Ceratopogonidae
	Gammaridae
	Crangonyctidae
	Cambaridae
	Coenagrionidae
Mason Neck NWR Great Marsh North	Belastomatidae
	Elmidae
	Chironomidae
	Ceratopogonidae
	Syrphidae
	Oligocheates
	Corixidae
	Hyalellidae
	Lestidae
	Polycentropodidae
Mason Neck NWR Great Marsh East	Belastomatidae
	Syrphidae

	Psychodidae
	Oligocheates
	Gammaridae
	Chironomidae
	Ceratopogonidae
	Hirudinidae
	Coenagrionidae
Mason Neck State Park East	Belastomatidae
	Hirudinidae
	Asellidae
	Corixidae
	Caenidae
	Elmidae
	Stratiomyidae
	Chironomidae
	Oligocheates
	Gammaridae
	Lestidae
	Crangonyctidae
	Hyalellidae
Mason Neck State Park Central	Belastomatidae
	Ephydridae
	Chironomidae
	Oligocheates
	Tipulidae
	Corixidae
	Elmidae
	Pontoporeiidae
	Gammaridae
	Caenidae
	Syrphidae
	Circulionidae
	Hydrophilidae
Mason Neck State Park West	Belastomatidae
	Chironomidae
	Caenidae
	Aeshnidae
	Libellulidae
	Hyalellidae
	Gammaridae
	Pontoporeiidae
	Crangonyctidae
	Syrphidae

	Haliplidae
	Hydrophilidae
	Ephydridae
Mattawoman Creek West	Belastomatidae
	Syrphidae
	Chironomidae
	Oligocheates
	Corixidae
	Gammaridae
	Aeshnidae
	Caenidae
	Lestidae
	Hyalellidae
Mattawoman Creek Island	Belastomatidae
	Oligocheates
	Chironomidae
	Haliplidae
	Stratiomyidae
	Ceratopogonidae
	Caenidae
	Lestidae
	Gammaridae
Mattawoman Creek Central	Gammaridae
	Oligocheates
	Chironomidae
	Lestidae
	Haliplidae
	Pontoporeiidae
	Hyalellidae
	Asellidae
	Elmidae
	Crangonyctidae
Mattawoman Creek North	Belastomatidae
	Libellulidae
	Syrphidae
	Oligocheates
	Pontoporeiidae
	Hyalellidae
	Chironomidae
	Corixidae
	Hirudinidae
	Ceratopogonidae
	Tabanidae

	Crangonyctidae
Mattawoman Creek East	Belastomatidae
	Stratiomyidae
	Chironomidae
	Corixidae
	Caenidae
	Lestidae
	Hyalellidae
	Gammaridae
	Haliplidae
Neabsco Creek West	Chironomidae
	Oligocheates
	Hirudinidae
	Tipulidae
	Gammaridae
	Ceratopogonidae
	Corixidae
	Lestidae
	Aeshnidae
Neabsco Creek North	Oligocheates
	Chironomidae
Neabsco Creek Central	Chironomidae
	Belastomatidae
	Oligocheates
	Ceratopogonidae
	Syrphidae
	Cambaridae
Neabsco Creek South	Belastomatidae
	Hirudinidae
	Ephydridae
	Empididae
	Dixidae
	Oligocheates
	Chironomidae
	Syrphidae
	Corixidae
	Hyalellidae
	Aeshnidae
Occoquan Bay Wildlife Drive	Asellidae
	Elmidae
	Libellulidae
	Chironomidae
	Ceratopogonidae

	Culcidae
	Oligocheates
	Tipulidae
	Corixidae
	Belastomatidae
	Ptychopteridae
	Polycentropodidae
	Lestidae
	Gammaridae
	Hydrachnida
Occoquan Bay Deephole Point North	Belastomatidae
	Elmidae
	Oligocheates
	Gammaridae
	Tipulidae
	Hirudinidae
	Chironomidae
	Asellidae
	Aeshnidae
	Cambaridae
	Stratiomyidae
Occoquan Bay Deephole Point South	Hirudinidae
	Oligocheates
	Chironomidae
	Palaemonidae
	Coenagrionidae
	Gammaridae
	Crangonyctidae
	Hyalellidae
Occoquan Bay Marumsco Creek	Oligocheates
	Chironomidae
	Corixidae
	Asellidae
Patuxent Refuge Rogue Harbor	Cambaridae
	Chironomidae
	Haliplidae
	Ceratopogonidae
	Tipulidae
	Libellulidae
	Culcidae
	Crangonyctidae
	Hyalellidae
	Coenagrionidae

	Lestidae
Patuxent Refuge Uhler Marsh	Chironomidae
	Gammaridae
	Cambaridae
	Lestidae
	Notonectidae
	Corixidae
	Culcidae
	Dixidae
Patuxent Refuge Knowles North	Hydrophilidae
	Belastomatidae
	Chironomidae
	Lestidae
	Baetidae
	Nepidae
	Culcidae
	Corixidae
	Elmidae
	Aeshnidae
	Gammaridae
	Palaemonidae
	Polycentropodidae
	Caenidae
Patuxent Refuge Knowles South	Chironomidae
	Nepidae
	Corixidae
	Hydrophilidae
	Caenidae
	Lestidae
	Elmidae
	Gammaridae
	Baetidae
	Belastomatidae
	Hirudinidae
	Asellidae
	Ephydridae
	Crangonyctidae
	Hyalellidae
Patuxent Refuge Lake Redington	Belastomatidae
	Oligocheates
	Corixidae
	Chironomidae
	Elmidae

	Culcidae
	Haliplidae
	Hydrophilidae
	Lestidae
	Baetidae
Powells Creek East	Belastomatidae
	Hirudinidae
	Chironomidae
	Gammaridae
	Asellidae
	Ceratopogonidae
	Oligocheates
	Corydalidae
	Curculionidae
	Coenagrionidae
Powells Creek Central	Hirudinidae
	Belastomatidae
	Corixidae
	Chironomidae
	Tipulidae
	Oligocheates
	Syrphidae
	Lestidae
Powells Creek South	Belastomatidae
	Chironomidae
	Oligocheates
	Asellidae
	Curculionidae
	Corixidae
	Polycentropodidae
Powells Creek West	Belastomatidae
	Asellidae
	Oligocheates
	Syrphidae
	Chironomidae
	Ceratopogonidae
	Hirudinidae
	Hyalellidae
River Terrace Fringe	Syrphidae
	Belastomatidae
	Elmidae
	Oligocheates
	Tipulidae

	Dysticidae
	Asellidae
	Gammaridae
	Hirudinidae
	Chironomidae

Appendix D: Summary of Variables Eliminated from Inclusion in Occupancy Modeling

Variable	Reason for Exclusion	Mean (St .Dev.)	Range
Plant species richness	Correlation coefficient >0.7	21.2 (7.6)	6 – 38
Percent cover perennial plant species	Correlation coefficient >0.7	115.4 (25.6)	35.7 – 194.0
Percent cover annual plant species	Correlation coefficient >0.7	23.8 (18.9)	0.05 – 75.4
Invertebrate family richness	Correlation coefficient >0.7	6.1 (2.1)	2 – 11
Shannon Diversity Index of invertebrates	Correlation coefficient >0.7	1.3 (0.36)	0.59 – 1.85
Marsh area (ha)	Not justifiable from literature	70.8 (80.3)	1.2 – 274.3

Appendix E: FQAI and CRAM Scores for Bird Survey Points

Point Name	CRAM score	FQAI
Dyke Marsh North	83.5	51.565
Dyke Marsh Lagoon	83.5	52.395
Dyke Marsh West	85	54.57
Dyke Marsh South	88	53.21
Featherstone NWR South	78.5	52.45
Featherstone NWR Central	77.5	46.345
Featherstone NWR North	79.25	49.86
Featherstone NWR Creek	85.5	51.135
Fort Belvoir Jack Abbott	76.25	44.635
Fort Belvoir Loop Trail	83	44.97
Fort Belvoir Wetlands Trail	90.5	50.185
Fort Belvoir Accotink Marsh	74	52.73
Fort Belvoir Accotink Bay	78	50.42
Huntley Meadows Marsh Inlet	87.75	41.185
Huntley Meadows Central Marsh	89.5	41.93
Huntley Meadows Creek East	82.5	40.785
Huntley Meadows Creek West	85.5	42.2
Kenilworth Gardens River Trail	63.5	50.64
Kenilworth Gardens Boardwalk	75.5	50.95
Kingman Marsh South	66.75	55.83
Kingman Marsh North	65.5	45.675
Mason Neck NWR Great Marsh West	97.5	48.82
Mason Neck NWR Great Marsh Central	97	55.585
Mason Neck NWR Great Marsh North	84.5	59
Mason Neck NWR Great Marsh East	88.5	54.815
Mason Neck State Park East	87	49.6
Mason Neck State Park Central	89	52.44
Mason Neck State Park West	80.5	54.065
Mattawoman Creek West	86	58.545
Mattawoman Creek Island	83.5	55.66
Mattawoman Creek Central	82	53.395
Mattawoman Creek North	86.5	52.46
Mattawoman Creek East	85	52.055
Neabsco Creek West	86.5	40.83
Neabsco Creek North	81	46.58
Neabsco Creek Central	80.25	55.605
Neabsco Creek South	85	65
Occoquan Bay Wildlife Drive	74	50.065
Occoquan Bay Deephole Point North	81	48.735

Occoquan Bay Deephole Point South	86.5	55.09
Occoquan Bay Marumsco Creek	80.5	50.555
Patuxent Refuge Rogue Harbor	86	41.63
Patuxent Refuge Uhler Marsh	70.5	44.835
Patuxent Refuge Knowles North	74.5	45.555
Patuxent Refuge Knowles South	80.5	51.42
Patuxent Refuge Lake Redington	65.5	45.44
Powells Creek East	83.25	49.235
Powells Creek Central	87	57.585
Powells Creek South	85.5	55.765
Powells Creek West	88	53.965
River Terrace Fringe	73	46.345

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